FINAL REPORT

AES HUNTINGTON BEACH GENERATING STATION SURF ZONE WATER QUALITY STUDY

Prepared for:
AES Huntington Beach
for submission to the
California Energy Commission

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EXECUTIVE SUMMARY

As part of the retooling of the Applied Energy Services (AES) Huntington Beach Generating Station (AES HBGS) in 2001, a near shore beach contamination investigation was required by the California Energy Commission (CEC). AB 411[1] exceedances on the beach have occurred with greater frequency in the area of the AES HBGS than at other locations on the beach. The CEC commissioned the study in July 2002 to specifically investigate the impact of the AES HBGS on recent ocean postings and closures. The objective of this study was to assess whether the operation of the AES HBGS was negatively impacting water quality in the surf zone in the vicinity of Huntington State and City Beaches. This report summarizes the investigation conducted by Komex on behalf of the CEC during the summer of 2002.

The scope of work consisted of: establishing a sampling and monitoring plan for a period of 14 weeks; sampling surface water and in-plant water daily for 14 weeks at eight sample locations, with sampling every three hours during a two-week intensive period from four of the eight locations; analyzing the water samples for AB 411 microbial indicators (total and fecal coliform, and *Enterococcus*) and physical and chemical parameters (pH, temperature, salinity, conductivity, DO, turbidity, and ammonia); conducting bacterial source tracking on selected samples collected during the intensive sampling period; performing an in-plant dye study to check for cross connections of the sanitary sewer system within AES HBGS; performing an outfall dye study to model thermal dispersion from the AES HBGS discharge pipe and to quantitatively predict the potential impact of the AES HBGS on the beach; monitor temperature and salinity profiles of ocean water and in-plant water using high sensitivity in situ sensors and data loggers; and to prepare a draft and final report for the CEC and AES HBGS.

The scope of work was designed to address four key questions relating to the potential impact of the AES HBGS on surf zone water quality:

- Is sub-thermocline freshwater drawn into the AES HBGS intake, or entrained into the thermal discharge of the outfall pipe?
- Are bacteria introduced into the AES HBGS intake from the ocean (at low concentrations), and selectively cultured by the elevated temperature of the cooling water in the plant?

[1] Assembly Bill 411 required routine sampling for three indicator bacteria in recreational waters used by more than 50,000 visitors per year; criteria levels were set at Total coliform 10,000 [MPN]/100 milliliters [ml]; Fecal coliform 400 MPN/100 ml; and Enterococcus 104 colony forming units [CFU]/100 ml)

- Are land-based sources of bacteria entering the discharge vault in the AES HBGS and are discharged to the ocean causing bacterial contamination of the surf zone of Huntington State Beach; and
- Are sanitary sewers within the AES HBGS connected to, or leaking into, the discharge vault?

In excess of 4,500 microbiological water quality samples were collected during the duration of the study. Samples were collected from eight locations within and adjacent to the AES HBGS; additional storm drain locations; beach and ocean locations. The principal sources of bacteria were Blackford's Ditch, the boiler fireside wash and to a lesser extent the boiler sump wash. Microbial water quality at these locations consistently exceeded AB 411 recreational water quality standards; bacterial concentrations within the two other on-site locations (general purpose retention basin and storm water sump) were lower.

Summary of Bacterial Exceedances During Daily and Intensive Sampling

DATE	LOCATION	BACTERIAL EXCEEDANCE
8/7/2002	DV0	Total Coliform: Fecal Coliform
8/23/2002	DV0	Enterococcus
9/2/2002	IV	Enterococcus
08/15/02e	DV0	Enterococcus
08/15/2003e	IV	Enterococcus
08/16/2002g	IV	Enterococcus
08/17/2002a	DV0	Enterococcus
08/17/2003g	IV	Enterococcus
08/18/2002g	DV0	Enterococcus
08/18/2002g	IV	Enterococcus
08/19/2002g	DV0	Enterococcus
08/19/2002g	DV10	Enterococcus
08/20/2002a	IV	Enterococcus
08/20/2002g	DV0	Enterococcus
08/20/2002g	IV	Enterococcus
08/21/2002a	IV	Enterococcus
08/22/2002g	IV	Enterococcus
08/23/2002g	DV10	Enterococcus
08/25/2002b	DV0	Enterococcus

Note:
IV: Intake Vault; DV0 Discharge vault at 0 feet; DV10 Discharge Vault at 10 feet; and a through H after the date represent three hour segments of a 24-hour sampling period, a being 0:00 to 3:00 hours through to h at 21:00 to 24:00

Bacterial concentrations at the intake vault and discharge vault were consistently low and exceeded AB 411 21 times (out of a total of approximately 1,700 samples).

The bacterial source tracking techniques used in this investigation, were able to differentiate the source of bacteria as either human (positive) or non-human (negative). Of the *Bacteroides* and *Prevotella* bacterial source tracking samples collected (64), two were confirmed as positive (human source); one from the Intake Vault on August21 at 9:40 PM (very near high tide), and the Discharge Vault on August 22 at 10:05 PM (also near high tide). Parallel measurements (not funded by this project) showed multiple positive results for this assay during the evening high tide of August 21, 2002 at locations at the Santa Ana River, Talbert Marsh, 6N, 9N, 12 N, 15N, and 18N sites. Of the Enterovirus bacterial source tracking samples collected (59), only two tested positive (human source-intake vault at 3:10 on August 21, 2002). Consequently, the AES HBGS was not a significant source of human-derived contamination during this study period.

A dye study was conducted on the restrooms within the AES HBGS to determine if they were leaking into the industrial wastewater discharge system. During the 24-hour sanitary sewer dye test, no dye was detected in the discharge vault with an in-situ fluorometer nor was any observed in either the discharge vault or the general purpose retention basin.

A dye study was performed on the AES HBGS discharge and within the near vicinity of the AES HBGS outfall, and the results incorporated into a thermal plume model. The thermal plume model has been used to determine the impact of bacteria discharged from the AES HBGS on surf zone water quality at 9N. Bacteria from the AES HBGS discharged to the ocean would account for <0.4% [total coliform]; <3% [fecal coliform]; and <12% [enterococcus] of the corresponding bacterial concentration on the beach at 9N.

Moorings were placed in the vicinity of the AES HBGS intake with an array of temperature, salinity, pressure and current sensors. Temperature, pressure and salinity sensors were placed in the intake vault and discharge vault at the AES HBGS. Investigations were performed on Blackford's Ditch and a storm water-sampling event occurred during the first significant storm of the season in November. No significant sub-thermocline water was drawn into the intake during the August period. However, some portions of the August temperature-salinity (T-S) data demonstrated warmer lower salinity water coming into the intake vault. During September, three cold-water events occurred at the intake mooring (water temperature less than 13.5°C), indicative of sub-thermocline water. Corresponding temperatures in the intake vault were lowered, suggesting partial entrainment of thermocline/sub-thermocline water by the AES HBGS.

Based on the results collected in this study the following conclusions have been made:

- One event in August and three brief events in September demonstrated entrainment of thermocline water into the intake vault, consequently thermocline water was occasionally entrained into the AES HBGS intake. During this study there were no specific indications that the intake water contained part of the OCSD plume;
- Concentrations of bacteria introduced to the AES HBGS from the ocean did not increase significantly in numbers during passage through the cooling water system prior to discharge;
- Land-based sources of bacteria (particularly Blackford's Ditch and Boiler Fireside Wash) did enter the discharge vault and were discharged to the ocean, but were not at concentrations high enough to contribute significantly to bacterial contamination of the surf zone of Huntington State Beach; and
- The sanitary sewers within the HBGS were not connected to, or leaking into the discharge vault.

Because of the microbial and oceanographic results presented in this study further investigation of the role of the AES HBGS facility as either a source, or as a thermocline/sub-thermocline transport mechanism to the coastal environment is not recommended. However the AES HBGS may play a role in the transport of bacteria from non-AES HBGS sources (local rivers and marshes for example) to focused sections of the Huntington State and City Beaches, further investigation of the thermal-plume onshore transport (T-POT) mechanism is warranted at this time. The T-POT mechanism may be facilitating the shoreward transport of contamination being transported up or downcoast through the surfzone.

As a general principle, discharges of fresh-water through the AES HBGS discharge vault and outfall should be minimized. Any revisions to the discharge limits for the AES HBGS should be determined as a result of a complete review of all available studies including this report during the NPDES permit renewal process (2004-2005). Additional recommendations include diverting the storm drains to prevent discharge of potentially contaminated urban runoff through AES HBGS, and treating runoff within the AES HBGS to remove contaminants (including heavy metals and bacteria).

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TERMS AND ABBREVIATIONS

AB	Assembly Bill
AES	Applied Energy Services
Ag	silver
AgCl	silver chloride
ANOVA	analysis of variance
ASTM	American Society for Testing and Materials
atm	atmospheres
BD	Blackford's Ditch
BFW	boiler fireside wash
bgs	below ground surface
BOD	biological oxygen demand
BSW	boiler sump wash
C	Celsius/Centigrade
CAO	Cleanup and Abatement Order
CEC	California Energy Commission
CFU	colony forming unit
cm	centimeters
cm/s	centimeters per second
COCF	chain-of-custody-form
COD	chemical oxygen demand
Ct	bacteria concentration at time t
CTC	5-cyano-2,3-ditolyltetrazolium chloride
°	degrees
DGPS	differential global positioning system
DHS	Department of Health Services
DNA	deoxyribonucleic acid
DO	dissolved oxygen
DOC	dissolved organic carbon
DV_0	discharge vault zero feet
DV_{10}	discharge vault ten feet
DV_{30}	discharge vault 30 feet
DVC	Direct Viable Count
DWR	California Department of Water Resources
EDI	electrical deionization

ELAP	Environmental Laboratory Accreditation Program
EMB	eosin-methylene blue
F	Fahrenheit
FS	fecal streptococcus
GIS	Geographical Information System
gpm	gallons per minute
GPS	Global Positioning System
GP	General Purpose Retention Basin
AES HBGS	Huntington Beach Generating Station
HBPS	Huntington Beach Purchase Sewer
Hz	Hertz
ID	internal diameter
IV	intake vault
K^+	Potassium ion
kg	kilograms
km	kilometer
L	Liter
LUSTIS	leaking underground storage tank information services
m	meters
M	molar
m^2	meter squared
m^{-2}	per meter squared
mg/kg	milligram per kilogram
mg/L	milligram per liter
mM	millimolar
m/s	meters per second
MJ	megajoule
ml	milliliters
MBC	MBC Applied Environmental Sciences
MPN	most probable number
MSB	male-specific bacteriophage
MSL	mean sea level
µg	micrograms
µg/kg	micrograms per kilogram
mg	milligrams
mg/kg	milligrams per kilogram

mg/l	milligrams per Liter
mm	millimeters
mS/cm	millisiemans per centimeter
ms/l	micrograms per Liter
MPN	most probable number
MW	monitoring well
Na^+	Sodium ion
NaCl	Sodium Chloride
ND	not detected
ng	nanograms
nm	nanometers
NPDES	National Pollutant Discharge Elimination System
NPDWR	National Primary Drinking Water Regulations
NRC	National Research Council
NWRI	National Water Research Institute
OCHCA	Orange County Health Care Agency
OCSD	Orange County Sanitation District
OCWD	Orange County Water District
%	percent
PCH	Pacific Coast Highway
PCR	polymerase chain reaction
PFU	plaque forming units
pg	picogram
ppb	parts per billion
psi	pounds per square inch
PVC	polyvinyl chloride
QA	quality assurance
QC	quality control
RNA	ribonucleic acid
RO	reverse osmosis
RT	reverse transcriptase
RT-PCR	reverse transcriptase polymerase chain reaction
RWQCB	Regional Water Quality Control Board Santa Ana Region 8
ΔSalinity	change in salinity
SCE	Southern California Edison
Spp.	species

State Parks	California State Department of Parks and Recreation
SWS	storm water sump
T(90)	decay coefficient
T/S	temperature and salinity
TAG	Technical Advisory Group
TC	total count
TDS	total dissolved solids
TKN	total Kjeldahl nitrogen
TOC	total organic carbon
TS	total solids
T/S	temperature salinity
UCI	University of California at Irvine
µg/L	microgram per liter
µm	micrometers
USC	University of Southern California
USEPA	U.S. Environmental Protection Agency
UV	ultraviolet
UV-B	ultraviolet band B
UV-C	ultraviolet band C
VBNC	viable but non-culturable
VOC	volatile organic carbon
W	watt
w/v	percentage weight by volume
w/w	percentage weight by weight
WAAS	Wide Area Augmentation System
ZID	zone of initial dilution

1 INTRODUCTION

The objective of this study was to assess whether the AES HBGS was negatively impacting water quality in the surf zone of Huntington State and City Beaches. As part of the agreement to retool two of AES HBGS' four power generating units in 2001, a near shore beach contamination investigation was required. The CEC commissioned the study in July 2002 to specifically investigate the impact of the operation of the AES HBGS on recent ocean postings and closures. This report summarizes the investigation conducted by Komex during the summer of 2002.

The report has been structured as follows:

- **Introduction.** The introduction includes information on the background of the project, study region, near shore ocean environment, study aims, and the scope of work;
- **Historical Summary**. This section provides a summary of previous investigations at the AES HBGS facility;
- **Methodology**. Methodology used in this investigation is described in detail in this section;
- **Results.** The results section presents the results obtained from long-term and intensive sampling within and adjacent to the AES HBGS, storm water sampling, the storm drain investigation, in-plant dye study, intake and outfall assessment, thermal plume model, offshore dye-study, and bacterial source tracking analysis;
- **Discussion.** The discussion section provides an interpretation of the results presented in the previous section;
- **Conclusions.** The conclusion concisely summarizes the key points presented in the discussion;
- **Recommendations.** The recommendations section lists several actions that can be undertaken to improve local surface and surf zone water quality;
- **Closure/Limitations.** The closure and limitations section describes the inherent limitations in the investigation;
- **References.** The reference section provides a bibliographic list of all references cited in the text of the report;
- **Appendix A.** A review of bacteria in coastal waters. The review describes the prevalence, persistence and significance of bacteria in coastal waters; and

- **Appendix B-J.** Include the daily field and sampling forms, site photographic documentation, chain-of-custody forms, thermal modeling parameters, laboratory reports and statistical tables.

The short section of beach that constitutes Huntington State Beach in Orange County has become one of the most heavily researched stretches of beaches in the U.S. Providing definitive answers to the causes of beach contamination in this region has been challenging and has often required multi-disciplinary teams of oceanographers, engineers, climatologists and microbiologists. In this study, four potential mechanisms by which the AES HBGS may impact local surf zone water quality have been clearly identified. Each mechanism has been outlined below, and established as a hypothesis. Each hypothesis has been tested and either disproved or accepted; if disproved the Alternate Hypothesis (*i.e.* the opposite) has been accepted.

Hypothesis 1: Sub-thermocline Entrainment

Is sub-thermocline freshwater entrained into the AES HBGS intake, or the thermal discharge of the outfall pipe?

Hypothesis 2: Plant Bacterial Proliferation

Are bacteria introduced into the AES HBGS intake from the ocean (at low concentrations), and selectively cultured by the elevated temperature of the cooling water in the plant?

Hypothesis 3: Land-Based Sources

Are land-based sources of bacteria entering the discharge vault in the AES HBGS and are discharged to the ocean causing bacterial contamination of the surf zone of Huntington State Beach; and

Hypothesis 4: Sanitary Sewers

Are sanitary sewers within the AES HBGS connected to, or leaking into, the discharge vault?

1.1 BACKGROUND

1.1.1 ASSEMBLY BILL 411

On July 26, 1999, in accordance with Health and Safety Code §115880 (Assembly Bill 411, Statutes of 1997, Chapter 765), the California Department of Health Services (DHS) was required to expand its regulations for ocean beaches (DHS, 1999). AB 411 and accompanying regulations contain weekly monitoring requirements, as well as requirements for ocean posting and closure. The requirements specifically apply to coastal beaches that are visited by 50,000 or

more people annually and are located adjacent to storm drains that flow from the period of April 1 through October 31. An estimated 550 million people visit California's public beaches annually contributing an economic value estimated to be over 27 billion dollars (State of California, 1999).

The Orange County Health Care Agency (OCHCA) and the Orange County Sanitation District (OCSD) collect water samples along the shoreline of Orange County for bacterial analysis. In the Huntington Beach area, the OCSD samples the beach surf zone up to five times per week at several locations to the north and south of the mouth of the Santa Ana River. The stations are sampled by OCSD and the data is reviewed by the OCHCA. If the data indicate that health care standards are exceeded, warning or closure signs are posted at the beach. Ocean water recreational closures are expected to occur when health risks (associated with recreational contact with ocean water) are considered greater than those associated with posting.

AB 411 stipulates monitoring for total coliform, fecal coliform, and the Enterococcus group of bacteria. In addition, the DHS has established seven standards (limits) for indicator bacteria concentrations which, if exceeded, require action on the part of the local health officer in the form of ocean postings. These postings inform beachgoers of recent or chronic elevated concentrations of indicator bacteria or close the water to recreational contact if the high levels of indicator bacteria are known to originate from sewage contamination (ocean closure).

The AB 411 standards and the national guidelines for marine water quality criteria published by the U.S. Environmental Protection Agency (USEPA, 1986) use epidemiological evidence to establish risk levels for swimming-associated illnesses based on the level of indicator bacteria, especially Enterococcus (State of California, 1999). The origin of Enterococcus in these epidemiological studies was presumed to be anthropogenic sources of fecal pollution including sewage, agricultural runoff, or urban runoff (McBride *et al.* 1998; Chang *et al.* 1990; Balarajan *et al.*, 1991; Corbett *et al.*, 1993; Haile *et al.*, 1999; Kay *et al.*, 1994; and von Schirnding *et al.*, 1992).

At the time of the investigation (Summer 2002), the OCSD discharged a blend of primary and secondary treated sewage through its outfall located approximately 5 miles southwest of the AES HBGS. The OCSD has been monitoring coastal water quality in the Huntington Beach area since 1958. The OCSD discharge practices (depth of discharge at 60 meters [m] [200 feet] the distance from shore of discharge at 7 kilometers [km] [4.4 miles], and high initial mixing ratio at 180:1) are aimed at reducing negative human-health-related impacts outside the zone of initial dilution (ZID). In addition, seasonal water column stratification (thermocline development) restricts the discharge plume to depths of 5 to 10 m (15 to 30 feet), and the high natural mixing

and predominant alongshore currents minimize potential for human risk (OCSD, 2002). Annual ocean monitoring and routine water quality monitoring indicate no significant environmental or human-health-related water quality effects due to discharge from the OCSD from 1985 to 1999 (OCSD, 2002).

Water Quality Criteria

Beach posting is recommended (and required for certain beaches under Health and Safety Code Section 115915 and the AB 411 implementing regulations) when indicator organism concentrations exceed any of the following levels for a single water sample:

- Total coliform 10,000 most probable number (MPN)/100 milliliters (ml)
- Fecal coliform 400 MPN/100 ml
- Enterococcus 104 colony forming units (CFU)/100 ml

Additional sanitary surveys and other related evaluations are recommended when indicator organisms exceed any of the following concentrations, based on the log-mean of at least five equally spaced samples in any 30-day period:

- Total coliform 1,000 MPN/100 ml
- Fecal coliform 200 MPN/100 ml
- Enterococcus 35 CFU/100 ml

Prior to the implementation of AB 411 and the new regulations, monitoring data from the OCSD demonstrated that indicator bacteria concentrations in the near shore waters off Huntington Beach were above normal in April 1999.

1.1.2 HUNTINGTON STATE BEACH—THE SUMMER OF 1999

On June 27, 1999, the concentration of total coliform exceeded the health standards (10,000 MPN/100 ml), and the first section of beach was closed on July 1, 1999 (OCSD 1999). The first ocean water recreational closure extended approximately 5,000 feet (1,520 m) along the Huntington State Beach between Newland Street and Magnolia Street. The affected area continued to spread to the south and resulted in two additional closures on August 6 and August 11, 1999. The bacterial plume then spread to the north, resulting in an ocean water recreational closure on August 18, 1999, which extended to Beach Boulevard. On August 20, 1999, the last remaining section of the Huntington State Beach south of Brookhurst Street was closed; the closure extended down to the mouth of the Santa Ana River. Two further ocean water recreational closures occurred at Huntington City Beach on August 24 and August 31,

1999, both extending northward from Beach Boulevard to their maximum extent at Goldenwest Street. In total, approximately 4.5 miles (7.2 km) of beach were closed due to bacterial contamination, extending north from the mouth of the Santa Ana River to Goldenwest Street.

Initial investigations to evaluate the source of the elevated bacterial concentrations were led by the OCSD with the participation of the OCHCA, the California State Department of Parks and Recreation (State Parks), the City of Huntington Beach, and the Regional Water Quality Control Board, Santa Ana Region 8 (OCSD, 1999).

OCSD reports some indication that the beach closures were more closely related to the implementation of AB 411's more stringent standards than an actual increase in bacterial concentrations at the beach (OCSD, 1999). An analysis of the monitoring data for 1998 indicated that if AB 411 standards had been applied in 1998, the OCSD-monitored beaches would have exceeded the single–sample water quality samples 143 times instead of the 12 occurrences under the previous standards. Monitoring data from April 1999 through September 1999 showed concentrations of indicator bacteria above normal, regardless of the incidence of beach closures. Sewage was initially suspected as the source since the signature of indicator bacteria reflected that of sewage (all three indicator bacteria showed elevations, and the ratio of fecal coliform to total coliform was 1:1). Later (August 1999), the level of both coliform concentrations declined, but that of Enterococcus remained high, a signature more similar to urban runoff.

Persistently high levels of all three-indicator bacteria in the surf zone between the Santa Ana River and the Huntington Beach Pier in 1999 prompted concern that the bacterial pollution was from an unidentified sewage source. A series of investigations were initiated by the OCSD in an attempt to identify the bacterial source.

1.2 STUDY REGION

1.2.1 STUDY LOCATION

The study location is centered on the AES HBGS at the intersection of Newland Street and PCH in Huntington Beach, California **(Plate 1-1)**. The study area includes the adjacent storm drains on Newland Street, the PCH, and the wildlife sanctuary **(Plate 1-2)**. Water samples were collected from as far south as the Santa Ana River, as far north as the Huntington Pier, and 0.25 miles off the shoreline into the Pacific Ocean **(Plate 1-3)**.

The offshore study area lies within the San Pedro shelf, located within the Southern California Bight (SCB), a region that extends south from Point Conception to San Diego. The SCB is dominated by the offshore equator-ward flowing California Current system, and that portion of the California Current that turns inshore and pole-ward and recirculates within the SCB (Southern California Eddy). A pole-ward flowing undercurrent transporting warmer, saltier Equatorial water from the south dominates the inshore region. The offshore study area extended from Huntington Pier to the Santa Ana River.

The on-shore study area lies entirely within the Coastal Plain of Orange County. The central lowland of the Coastal Plain of Orange County stretches northwest from Irvine past Santa Ana and Garden Grove into Los Angeles County. It has little relief and an average slope of less than 20 feet per mile (3.8 meters per kilometer [m/km]) (Department of Water Resources [DWR], 1967). The Puente Hills, Santa Ana Mountains, and San Joaquin Hills bound the central lowland to the north, east, and southeast, respectively, and by the Pacific Ocean to the south and southwest (Morton *et al.,* 1976). The central lowland comprises the Downey and Tustin Plains with the Study Area lying on a coastal section of the Downey Plain. The plain formed from alluvial deposits carried by the Los Angeles, San Gabriel, and Santa Ana Rivers (DWR, 1967).

Along the coast of Orange County are low hills and mesas aligned northwesterly as a surface expression of the Newport–Inglewood structural zone. The low hills are mesas breached by stream-cut gaps and ancient meanderings of the Santa Ana River through which fingers of the central lowland extend to the sea (DWR, 1967; OCWD, 1999). The eastern boundary of the Study Area lies within one of these physiographic depressions known as the Talbert Gap.

To the northwest is Huntington Beach Mesa, which is approximately 2 miles (3.2 km) wide and extends inland nearly 4 miles (6.4 km) from the ocean. A cliff 30 to 40 feet (9.1 to 12.2 m) high abuts on the ocean. The surface of the mesa slopes inland and eventually submerges under the sediments of the Downey Plain. The majority of the Study Area is within the Huntington Beach Mesa with the exception of the eastern boundary as noted above.

1.2.2 SITE CHARACTERISTICS

1.2.2.1 Climate & Precipitation

General climatic characteristics provide insight as to the seasonal variability of the near shore salinity and temperature characteristics in the water column. The general climatic characteristics have been described as follows:

- Mediterranean climate mild short winters, warm dry summers;
- 90% of precipitation between November and April;
- Summer temperature range 61 to 79 °F (average daily low – average daily high);
- Winter temperature range 51 to 69 °F (average daily low – average daily high); and
- Average summer temperature 70 °F;
- Average winter temperature 60 °F;
- Annual average maximum temperature 74.1 °F;
- Annual average minimum temperature 55.9 °F; and
- Annual precipitation of 13.7 inches.

Data derived from the Western Regional Climate Center, Los Angeles Civic Center, California (045115), data averaged over the time period January 1, 1914 to December 31, 2001.

1.2.2.2 Wind

Historical documentation summarizes the general wind climate to be as follows:

- Prevailing winds out of the northwest in the summer with sea breezes to 40 km/hour (25 miles per hour [mph]); and
- Winter winds generally offshore and from the southeast.

Wind stress on the surface of water causes waves, surficial mixing, and, depending on persistence, co-flowing surficial currents in the order of 2% to 3% of the wind speed in an upcoast or downcoast orientation. The shoreline is relatively straight and aligned in a northwest / southeast orientation. In summer, when northwest sea breezes occur, wind-driven surficial currents can be expected to be downcoast parallel to the beach at a distance of about 8 km offshore (Noble et al., 2003). Nearer the beach, the wind direction becomes more onshore during the afternoon resulting in onshore flow at the surface and subsurface offshore flow. Alternately, during the night the sea breeze reverses at the beach resulting in offshore flow at the surface and subsurface shoreward flow. During the winter, winds are from the southeast, generating wind-driven surface currents that are generally upcoast.

Relative to the AES HBGS outfall, a surfacing thermal plume can be driven laterally (co-flowing with the wind) at a rate of speed on the order of 0.22 to 0.33 meters per second (m/s) (0.7 to 1.0 feet/second) for a 40 km/hour (25 mph) wind.

1.2.2.3 Oceanography

The coastal ocean off of Huntington and Newport Beaches is typically stratified during summer. The water column is characterized by a surface mixed layer that is 5-10 meters thick, a strong seasonal pycnocline that extends from the base of the surface mixed layer to a depth of 20 to 40 meters, and a weaker permanent pycnocline below the seasonal pycnocline (Noble et al., 2003). The coastal currents are characterized by tidal fluctuations and remotely forced variability in the low frequency (subtidal) component of the flow. On average, currents over the shelf are downcoast (toward San Diego) during the summer within the upper layer extending from the surface down to a depth of about 25 meters. Over the continental slope below 70 meters depth, the currents are typically upcoast (toward Los Angeles). Large-scale current fluctuations over the shelf have a magnitude similar to the mean and are larger on the outer shelf than on the inner shelf. The time scale of fluctuations on the shelf is on the order of 7-20 days (Noble et al., 2003).

There are strong diurnal and semidiurnal contributions to temperature and currents over the shelf. Sea surface temperature routinely demonstrates an afternoon maximum. The daily sea breeze pattern creates an afternoon shoreward forcing, which is realized in the nearsurface currents with a compensating offshore flow subsurface. There is a corresponding nighttime reversal in these processes. Semidiurnal variability is also present over the shelf (Noble et al., 2003). Closer to shore the semidiurnal variability is expressed in cross-shelf oscillations of the flow that result in little net transport (Noble et al., 2003).

Within the surf zone currents are driven by the wave direction and magnitude. Typically, the waves impinging on the coastline are either from the west or south. Because of the geometry of the coastline, waves from the south will generate upcoast transport in the surf zone, and waves from the west will generate downcoast surf zone transport. Often waves from both directions are present, but the direction of flow within the surf zone will depend on the relative magnitude of the significant wave heights of waves from the two sources. Much of the time, the predominant waves are from the south and drive an upcoast flow during the summer, but downcoast flow in the near shore zone is also common.

Internal waves and tides cause oscillations of the pycnocline within the water column. Strong oscillations can reach all the way to the surf zone and cooling within the surf zone can result when there is a significant enough nearshore elevation of the pycnocline (Noble et al., 2003).

The tides in the study area are mixed, semi-diurnal with the characteristics identified in **Table 1-1**. Values referenced relative to Newport Beach, Newport Bay Entrance Tide Station, Station ID 9410580 (Latitude: 33d 36.2m N Longitude: 117d 53.0m W).

Tides

Table 1-1 Tidal Characteristics from the Newport Beach Tidal Station

Height (feet)	Condition	Description
7.86	Maximum	January 28, 1983
5.39	MHHW	Mean Higher High Water
4.65	MHW	Mean High Water
2.79	MSL	Mean Sea Level
2.76	NGVD29	National Geodetic Vertical Datum (1929)
0.93	MLW	Mean Low Water
0.37	NAVD88	North American Vertical Datum (1988)
0.00	MLLW	Mean Lower Low Water (Chart Datum)
-2.16	Minimum	January 20, 1988

The mean tidal range is 3.7 feet. The normal large tidal range is 5.4 feet. Flood tides typically add an upcoast or west-northwest component to the ambient flow and ebbing tides a downcoast or east-southeast component.

Currents

Local oceanographic conditions have been extensively researched and reviewed in previous investigations, particularly the first three phases of Huntington Beach investigations. The reports provide an accurate summary of local conditions, and do not need to be reviewed further. Historical documents note that currents in the near shore area are complicated by eddies, wind, weather, and tides. Data extracted from graphics in the report indicated the following current trends:

Table 1-2 Historic Tidal References

Period	Dominant Current Direction	Current Speed (feet/min)
January to March 1973	Southeast (parallel to shore)	25 (0.127 m/s)
April to June 1973	Southeast (parallel to shore)	15 (0.076 m/s)
July to September 1973	West (offshore at 45°)	50 (0.254 m/s)
October to December 1973	East (shoreward at 45°) Southeast (parallel to shore)	45 (0.229 m/s) 65 (0.332 m/s)
January to March 1974	North (shoreward at 45°) Southeast (parallel to shore)	20 (0.107 m/s) 12 (0.061 m/s)

Current meters were installed by USGS at two sites in the immediate vicinity of the AES HBGS outfall for the period of June 16 through October 10, 2001 **(Plate 1-5)**. The USGS data from 2001 indicated that mean current speeds ranged from 0.03 to 0.05 m/s, with the dominant current direction being approximately parallel to the shoreline. Direct shoreward currents (within 30 degrees to either side of the shortest distance to shore) occurred less than 15% of the time, with mean speeds of less than 0.05 m/s **(Plate 1-5**). Maximum nearsurface alongshore velocities from USGS mooring 3 (~1300 m offshore from the AES cooling water intake) ranged from more than 0.5 m/s downcoast to more than 0.5 cm/s upcoast. However, the mean nearsurface flow was downcoast.

1.2.3 GEOLOGIC SETTING

Geologically, the Study Area lies on the southwest limb of the Los Angeles Basin, within the Peninsular Ranges Province of California (Morton *et al.*, 1976). The Los Angeles Basin is a structural basin and lowland area bounded to the north by the Transverse Ranges and to the east by the Peninsular Ranges.

Directly beneath the Study Area, the Newport-Inglewood alignment of folds and faults forms the uplifted southwestern limb of the Basin. The Newport-Inglewood uplift extends for approximately 42 miles (67.6 km) in a northwest-southeast direction from Beverly Hills in Los Angeles County to Newport Beach in Orange County (DWR, 1967). In general, rock units dip down toward the ocean and landward from the axis of the faulted anticlinal feature of the

Newport-Inglewood Fault Zone. The surface expression of the uplifted belt is an alignment of low coastal hills and mesas, transected by several erosional features or gaps (DWR, 1967).

There has been considerable vertical movement along the length of the Newport-Inglewood structural zone with a southwest block of metamorphic rocks uplifted above the basement rocks on the northeast side. Increased displacement with depth suggests that there has been repeated movement along an established pattern. The Lower Pleistocene deposits have been displaced vertically by as much as 300 feet (91.4 m) and laterally by as much as 0.5 mile (800 m), with the seaward block moving northwest and the inland block moving southeast (DWR, 1967).

Deposits in the basin to the northeast represent Quaternary and Tertiary hybrid marine and continental sediments overlying a pre-Tertiary basement. North of the Study Area, where deposition is considered to have been continuous, these deposits have reached a thickness of more than 20,000 feet (6,096 m) since the Middle Miocene. Recent (Holocene) deposits extend from the Coastal Plain through the gaps to the ocean (DWR, 1967).

On the southwest basin margin, folding, faulting, and erosion associated with the Newport-Inglewood uplift has resulted in marked lateral variations in thickness and lithology along with numerous unconformities and stratigraphic discontinuities. In the Study Area, recent deposits unconformably overlie upper Pleistocene alluvial and terrace deposits (DWR, 1967).

1.2.4 REGIONAL HYDROGEOLOGY

The Orange County Groundwater Basin is located in the northern portion of Orange County, and is overlain by both the Tustin and Downey Plains. The basin covers an area of approximately 350 square miles (906 square km), and is bordered by the Coyote and Chino Hills to the north, the Santa Ana Mountains to the northeast, and the Pacific Ocean to the southwest. The basin extends to the Orange county line to the northwest, where its aquifers continue into the Central Basin of Los Angeles County. The southwestern boundary of the basin is represented by The Newport-Inglewood Fault Zone, which inhibits groundwater flow in all but the shallow aquifers (OCWD, 1999).

The aquifers within the Orange County groundwater basin can extend to depths of over 2,000 feet (610 m). They form an interconnected series of sand and gravel deposits, with interbedded clays and silts (DWR, 1967). The clays and silts are more predominant in the coastal regions, while inland, the deposits become thinner and increasingly discontinuous, allowing groundwater flow between the shallow and deeper aquifers (OCWD, 1999).

The Orange County Groundwater Basin was divided into two major subdivisions, the Forebay and Pressure Areas, by the DWR. The Forebay Area, the area of recharge, comprises basins replenished by direct percolation from surface waters or by vertical groundwater flow from overlying hydraulically connected aquifers. The Forebay Area encompasses most of the cities of Anaheim and Fullerton and portions of Orange. The Pressure Area comprises the basin area where surface water and shallow groundwater are prevented from percolating in large quantities into the major producing aquifers by shallow (upper 50 feet [15.2 m]) clay and silt layers (OCWD, 1999). Most of the central and coastal portions of the basin fall within the Pressure Area.

The Orange County Groundwater Basin consists of three major aquifer systems: the Shallow, the Principal, and the Lower Aquifers (OCWD, 1999). The Shallow Aquifer System comprises recent (Holocene) sediments deposited after the Late Pleistocene lowering of sea level. Erosion of the exposed landmass by streams and rivers spread clay; silt; sand; and gravel over large portions of the basin with recent deposits reaching up to 175 feet (53 m) thick near the coast (DWR, 1967). The lower section of these deposits is composed of interfingering lenses of coarse sand and gravel, which forms the Talbert Aquifer (DWR, 1967). The Talbert Aquifer extends from the Forebay area at the mouth of the Santa Ana Canyon and offshore south of the Newport-Inglewood Fault Zone.

The uppermost recent (Holocene) deposits are comprised of stream channel and unconsolidated and semi-consolidated alluvial fan and flood plain sediments and are in part argillaceous (clay or clayey), confining the Talbert Aquifer (DWR, 1967). Within the confining layer, lenses of silt, sand, and gravel form perched and semi-perched aquifers along the coast and in scattered sections of the main part of the basin.

The Talbert Aquifer extends from offshore to the Forebay area at the mouth of the Santa Ana Canyon (OCWD, 1999) and is the only aquifer in direct contact with the Pacific Ocean (National Research Council [NRC], 1994). Beneath the Talbert Aquifer, elevated blocks of the Newport-Inglewood fault zone inhibit groundwater movement between the central lowland and the ocean for all but the shallow aquifers (OCWD, 1999). The deeper aquifers are subject to seawater intrusion due to their contact with the Talbert Aquifer (NRC, 1994). Seawater intrusion was first observed in municipal wells during the 1930s because of basin overdraft. The overdraft continues, causing seawater to migrate up to 3.5 miles (5.6 km) inland (NRC, 1994). Despite the Talbert and Alamitos injection barriers, the OCWD (1999) Balanced Basin Hydrologic Budget (1996-1997), estimates seawater inflow to the basin at 2,500 acre-feet per year (afy) (3,083,717 cubic meters per year [m^3/y]). The OCWD estimates that flow provided by

the injection barriers are not lost to the ocean, and that the net inflow of seawater described above is occurring as shown in the Talbert Aquifer system.

Injection wells have been sited along Ellis Avenue, within the 2.5-mile (4.0 km) gap between the Huntington and Newport Mesas, located approximately 3.5 miles (5.6 km) from the ocean, in an attempt to reduce intrusion of seawater to the Talbert Aquifer. Water is injected into four aquifer zones along this barrier, using reclaimed water produced at Water Factory 21. Currently, the injection water is composed of 65% reclaimed water and 35% deep well water (OCWD, 1999); however, due to age and deterioration, the wells are functioning at a decreased capacity (OCWD, 1999). As a result, groundwater within the Talbert Aquifer appears to flow northeast (inland) approximately as far as Adams Avenue where there is a confluence with groundwater flowing from the north.

Groundwater flow directions in the heterogeneous sediments overlying the Talbert Aquifer appear to be generally inland, although groundwater flow direction are highly variable and locally appear vertically downwards, possibly providing recharge to lower perched aquifers or to the Talbert Aquifer. Information from the RWQCB Leaking Underground Storage Tank Information System (LUSTIS), collected for a property in Huntington Beach along PCH, indicates the presence of groundwater in two separate zones, each zone generally less than a few feet in thickness. Groundwater in the upper (shallow) zone was shown to occur beneath the property as perched groundwater under unconfined conditions, while groundwater in the lower zone is apparently confined and may be perched. Groundwater flow in the two zones was found to differ in direction and magnitude. In December 1992, groundwater was flowing east (inland) in the upper zone and northeast (inland) in the lower zone. Groundwater was described as brackish in the upper zone and saline in the lower zone, attributable to irrigation of grass and shrubbery in the area around a nearby hotel that may have resulted in reduced salinity in the upper zone (GTI, 1993).

1.2.5 THE AES HBGS

The AES HBGS is located at 21730 Newland Street, in the City of Huntington Beach in Orange County, California. It is situated approximately 1.5 miles (2.4 km) southeast of the center of the city of Huntington Beach at the intersection of PCH and Newland Street and is approximately 1,050 feet (320 m) from the shore of the Pacific Ocean **(Plate 1-1).**

The AES HBGS was constructed in the late 1950s and has five power generation units. Units 1, 2, 3, and 4 are boiler/steam turbine units, and Unit 5 is a peaker unit comprised of eight combustion turbines. Units 3 and 4 had been operated very sparingly after 1989 and were

retired in 1995 by Southern California Edison (SCE). AES HBGS purchased the power-generating facilities from SCE in 1998. Units 3 and 4 underwent retooling in 2001. Eight vertical-type circulating pumps (two for each condenser) provide ocean water to the steam turbine condensers and the closed-loop cooling system.

Pacific Ocean – Circulating Cooling Water Supply

Cooling water is supplied to the station from the ocean through a single 14-foot inner diameter (ID) concrete pipe located 1,650 feet (500m) offshore at a depth of 15.8 feet (4.8 m) off the ocean floor, and 11.7 feet (3.6m) below mean lower low water (MLLW), the maximum permitted intake velocity is 2.0 ft/s (0.61 m/s) (see **Figures A-1, A-2 and A-3**). The flow is directed to the intake forebay and a screening facility within the station. After passing through the screen system, the cooling water is pumped to the condensers, which raises the water temperature approximately 21.9°F when the station is operating at full capacity. After passing through the condensers, the water returns to the discharge vault. Each circulating pump has a flow rate of 44,000 gpm. The approximate circulating water flow rate for each condenser is about 88,000 gpm (333,116 L/min) and the maximum flow rate for the entire plant is approximately 352,000 gpm (1,332,465 L/min) (507 million gpd [1,919 million L/day]). The water is discharged to the ocean from the discharge vault through a single 21-foot internal diameter concrete pipe located 1,500 feet (450 m) and 25 feet deep (7.62 m) offshore (SCE and Kunz, 2002) at a theoretical maximum velocity of 5.1 ft/s.

The theoretical velocity assumes that there is no wear on the circulating pump impellers and that they are pumping at their rated discharge; and that there is not an accumulation of mollusks on the inside of the discharge pipe wall. If wear on the pump impellers has occurred, this will change the shape of the pump curve (discharge versus head). If mollusks have significantly accumulated on the inside of the discharge pipe wall, this will change the shape of the system curve (discharge versus head). The point at which the pump curve intersects the system curve is called the "operating point". This is where the system will operate and discharge water at the associated flow rate and head. If the location of the operating point changes (i.e. it's forced further up the pump curve as the pump impellers wear and/or mollusks accumulate on the inside of the discharge pipe wall), the discharge of the system will decrease and the required head to maintain that discharge will increase. As such, the actual flow velocity in the discharge pipe could be significantly different than the theoretical velocity noted above. Since discharge is equal to the velocity multiplied by the internal cross-sectional area of the pipe, the associated discharge could also be significantly different than the theoretical discharge

noted above. The only way to determine the actual velocities in the discharge pipe would be to install instrumentation to monitor the velocity and/or flow rate.

The discharge pipe releases the heated water vertically from a depth of approximately 11 feet (3.4 m) below the surface at MLLW. This depth is referenced from the top of the riser that extends approximately 10 feet (3.0 m) above the seabed.

The returned ocean water should not exceed the AES HBGS's current NPDES permit limits. Under normal operating conditions, the current NPDES permit allows a maximum of a 30 °F increase in the return ocean water temperature over the natural temperature of the receiving waters, as measured by the intake water temperature. With two units operating, the average discharge velocity is 1.1 ft/s.

Circulating Cooling Water Treatment

Water flow through the intake and cooling system is sustained with maintenance procedures that principally prevent biofouling. To control the growth of algae and bacteria, the circulating ocean water is periodically treated with sodium hypochlorite solution through the suction of each circulating pump. The chlorination treatment is performed for approximately 30 minutes at 12-hour intervals. Accumulated marine growth (primarily mollusks), is removed either by heat-treating the circulating water system supply or manual removal by divers.

Prior to the acquisition of Southern California Edison (SCE) Huntington Beach by AES, the units in operation were heat-treated every five to six weeks. A heat treatment involves pumping a portion of the water from the discharge vault back through the condensers for an additional temperature increase (up to approximately 105°F) and then discharging it out the intake conduit to kill marine-fouling organisms. Upon acquisition of the plant by AES HBGS, this process was eliminated, and alternative treatment methods for elimination of growth were tested; however, such alternate methods proved unsuccessful in reducing the marine growth. As such, AES HBGS is currently reviewing the process and has re-implemented a heat treatment procedure.

Between July 1998 and September 2002, heat treatments were conducted at the plant on the following dates: August 11, 2001; November 10, 2001; December 24, 2001; February 15, 2002; April 2, 2002; June 15, 2002; August 4, 2002; and September 15, 2002. Heat treatments were not conducted between July 1998 and July 2001 (Kunz, 2002).

Make-Up Water for Power/Steam Turbine Production

The boiler feed water treatment system is designed to service Units 1 through 4. Boiler feed water is supplied by the City of Huntington Beach. The raw water feed to the existing demineralizer system includes the following:

- Three 50% multi-media filters to remove suspended solids in order to meet the reverse osmosis (RO) system suspended solids requirements;
- Two 100% carbon filters to remove any chlorine residual from the city water;
- Two 100% primary and secondary water softener systems to remove hardness in the water in order to prevent scaling of the RO membranes;
- One ultraviolet (UV) sterilization unit to prevent biological growth;
- Three 33% RO booster feed pumps rated at 74 gpm and 400 pounds per square inch (psi) that feed a three-train RO treatment system designed to remove other dissolved ions in the softened water; and
- A four-stack electrical deionization (EDI) system to polish the treated RO effluent to demineralize make-up water quality.

The system is designed to produce approximately 120 gpm of high purity demineralized water. The RO reject water and EDI reject water are sent to a recycle tank. The recycle tank water is used as dilution water to make up the concentrated brine used for regeneration of the softener. Backwash and wastewater from the filters and softeners is discharged to the general purpose retention basin, which is part of the AES HBGS' wastewater system.

Water Treatment Requirement

Chemicals added to the circulating cooling water and service water systems are stored on site. The chemical feed system consists of tanks, pumps, piping, instrumentation, and controls for introducing chemicals into the feed water, condensate, and cooling water system to prevent corrosion. Hydrazine and ammonia are used to maintain the pH of the water within prescribed limits of 9.1 to 9.4 and to remove DO. Corrosion inhibitor is used to prevent the formation of scales in the cooling water system.

Wastewater Discharge

Several wastewater streams are generated during operation and maintenance of AES HBGS. The circulating cooling water is by far the largest volume. Other wastewater streams generated in small volumes include equipment wash water, floor-drain streams, blow down streams,

storm water, and sanitary waste. The unit sumps pump blower wash-down and infiltrating groundwater from the units into the retention basins (MBC, 2002). Sanitary wastewater is discharged to the City sewer system, while other wastewater streams (including storm water) flows are collected in onsite retention basins and subsequently discharged to the Pacific Ocean.

The east and west retention basins (also called the general purpose retention basins [GP]) are used as a bulk wastewater storage facility. The basins are located at the southern boundary of the site, adjacent to the wildlife sanctuary alongside the PCH **(Plate 1-2).** The basins receive water from yard and in-plant drains. The water from the basins is pumped into the discharge vault. The west retention basin well is a sump well adjacent to the west retention basin.

In addition, an oil/water separator treats any oily wastes from various drains throughout the AES HBGS. Oily sludge resulting from the separation process is disposed of offsite in a licensed/permitted facility. The resulting wastewater is discharged to the retention basin.

Wastewater resulting from periodic plant maintenance cycles may contain concentrations of heavy metals. Metal cleaning wastewater is collected and discharged to the retention basins.

In addition, offsite urban run-off and storm flows are directed to the discharge vault. The PCH/Newland storm drain is located south of the AES HBGS, on the northwest side of the parking lot of the Wetland Wildlife Care Center, and drains into the discharge vault (OCSD, 2002).

1.3 STUDY AIMS

The aims of this study are based around the four hypotheses and are as follows:

The Sub-Thermocline Hypothesis

Hypothesis 1, page 2

To assess whether sub-thermocline freshwater is drawn into the AES HBGS intake containing elevated bacteria concentrations, and/or sub-thermocline fresh water is entrained into the thermal discharge of the outfall pipe, facilitating the shoreward transport of bacteria. Analysis of the temperature and salinity profile of any incoming colder water will assist in identifying the source as either freshwater or colder ocean water. Analysis of the ammonia concentration in incoming water into the powerplant will also assist in clarifying the source of the colder water.

Plant Bacterial Proliferation

Hypothesis 2, page 2

Daily analysis of bacteria concentrations within the powerplant intake and discharge vaults; including round-the clock sampling over a two-week period, facilitate an assessment of the thermal effect of the cooling water on bacterial concentrations.

Land-Based Sources

Hypothesis 3, page 2

Daily samples of bacteria will be collected from onsite land-based (non-ocean) sources and offsite land-based sources to assess the potential contribution of these sample locations. Bacterial source tracking methodologies will also be employed to identify the source of any fecal matter identified during the daily sampling. The bacterial source tracking samples will be collected during the two-week intensive sampling period to maximize the value of the samples collected.

Sanitary Sewers

Hypothesis 4, page 2

A dye study will be conducted on the sanitary sewers within the AES HBGS to establish whether there has been a cross-connection to, or are leaking into, the discharge vault.

Oceanographic Environment

The thermal discharge from the AES HBGS will be evaluated during a series of dye studies forming the basis of an empirical dilution model that will be compared to theoretical computer modeling of the thermal plume. These tools will be used to assess bacterial transport mechanisms in the local oceanographic environment off Huntington State Beach. The models will assist in retrospectively assessing the impact of historical bacterial concentrations recorded in the AES HBGS in the summer of 2001.

1.4 SCOPE OF WORK

The scope of work consisted of the following tasks:

- Establish a Sampling and Monitoring Plan for a period of 14 weeks and identify eight appropriate sample locations;

- Sample surface water and in-plant water daily for 14 weeks at each of the eight sample locations, and sample every three hours during a two-week intensive period from four of the eight locations;
- Analyze the water samples for AB 411 microbial indicators (total and fecal coliform, and Enterococcus) and physical and chemical parameters (pH, temperature, salinity, conductivity, DO, turbidity, and ammonia);
- Conduct bacterial source tracking on selected samples collected during the intensive sampling period;
- Perform an in-plant dye study to check for cross connections of the sanitary sewer system within AES HBGS;
- Perform an outfall dye study to empirically model transport and dispersion of the plume from the AES HBGS discharge pipe and to quantitatively predict the potential impact of the AES HBGS discharge on beach water quality. Attempt to couple these empirical results from the dye study to theoretical predictions from computer dilution model;
- Monitor temperature and salinity profiles of ocean water and in-plant water using high sensitivity in situ sensors and data loggers; and
- Prepare draft and final reports for the California Energy Commission and AES HBGS.

2 HISTORICAL SUMMARY

2.1 PHASE I INVESTIGATIONS

- The first set of investigations (Phase I) took place between July and November 1999. The focus of these investigations was on sewage infrastructure in the area, with some sampling of groundwater and urban runoff. Possible sources of bacterial contamination were prioritized according to their potential to contain human fecal material and their potential concentration of fecal coliform bacteria. The Priority One sources identified (in order of priority) included the OCSD Coast Trunkline;
- OCSD Coast Trunkline Siphon;
- OCSD Discharge Plume;
- State Parks Restrooms;
- OCSD 120-inch diameter Outfall;
- OCSD 78-inch diameter Outfall;
- OCSD Huntington Beach Purchase Line; and
- OCSD Newland Line.

Priority Two potential sources included the City of Huntington Beach and County Pump Stations, possible unmapped pipe and conduit, OCSD Plant No.2, Talbert Channel, and the Santa Ana River. Priority Three and Four potential sources included the AES HBGS Outfall, the AES HBGS, and a construction site. Contamination transport mechanisms along the beach were also proposed and preliminary investigations conducted. No significant sewage leaks were discovered, and no contaminated groundwater was found by OCSD investigations. High levels of indicator bacteria were found in urban runoff, and the Santa Ana River and Talbert Marsh demonstrated as transport mechanisms.

The Phase I investigation was reviewed and critiqued in late February 2000, the efforts of the OCSD were commended and suggestions made for further investigations, that subsequently became a component of the Phase II investigations.

2.2 PHASE II INVESTIGATIONS

The Phase II investigations took place between December 1999 and July 2000 to address the more immediate investigative needs. The investigations addressed three major issues: the nature of surf zone water quality impairment at Huntington State Beach and Huntington City

Beach; the efficacy of current monitoring and health-advisory programs, and the potential sources of indicator bacteria and pathways by which these bacteria might be transported to the surf zone (OCSD, 2002).

The surfzone water quality investigations demonstrated that indicator bacteria were concentrated in a region of the surf zone bounded by the Santa Ana River to the south and the Huntington Beach Pier to the north during dry weather periods. During wet weather periods, elevated levels of indicator bacteria extended over a broader region of the surf zone both north and south of the Santa Ana River. The geometric mean of indicator bacteria in the surf zone at Huntington State Beach and Huntington City Beach was only marginally worse during the summer of 1999 compared to either the summer of 1998 or the summer of 2000. The statistical nature of bacterial monitoring data greatly influenced the incidence of beach closures because bacterial monitoring data tended to be lognormal and/or power-law distributed. Furthermore, because of the 24-hour lag between sampling and completion of data analysis, beach advisories were often issued at times when water quality may have satisfied applicable standards. Conversely, advisories may not have been in place when water quality failed to satisfy applicable standards.

A statistical analysis of the available water quality data and various environmental factors (*i.e.*, predominant winds, tide status, dominant currents, etc.) revealed that during the three summer periods for which complete data sets were available (1998, 1999, and 2000) surf zone water quality was generally worse during large spring tides, and surf zone water quality generally improved during neap tides. The relationship between the concentration of indicator bacteria and tidal range suggested that indicator bacteria entered the surf zone by tidal action. The statistical analysis also revealed that the concentration of indicator bacteria in the surf zone was significantly higher at night and lower during the day, which could influence sampling of the surf zone, routinely conducted early in the morning. The daily decrease in indicator bacteria concentrations during daylight hours appeared to be caused by sunlight-induced mortality (**Appendix A**).

Ocean current dynamics were assessed through the use of a series of dye studies. The dye study on the Talbert Marsh demonstrated that much of the water flowing from Talbert Watershed was entrained in the surf zone, then transported parallel to the shore. Waters move upcoast when swells are out of the south to southwest. Conversely, waters move downcoast when swells are out of the west to northwest. Some of the dye in the study traveled from the Talbert outlet offshore by rip currents. The dye study on the Santa Ana River revealed that

waters flowing out of the Santa Ana River passed through the surf zone and entered the region outside of the breakers.

Potential sources of surf zone pollution, as evaluated during the Phase II investigations, included the following:

- OCSD's wastewater outfall entrained in the AES HBGS outfall,;
- Subsurface sewer collection systems,;
- Nuisance runoff from the Talbert and Santa Ana River Watersheds due to upland irrigation overspray, which provides a significant reservoir of indicator bacteria that accumulates in pump station forebays before being discharged into a network of channels that drain to the beach; and
- Natural sources of indicator bacteria in the surf zone such as Talbert Marsh, along with, potentially, other coastal wetland systems in the area (*e.g.* Newport Slough along the Santa Ana River).

Phase II Investigations were inconclusive as to the role of storm water pump station diversions in averting bacterial contamination in the surf zone.

- In November 2000, a blue-ribbon panel of experts was impaneled by the National Water Research Institute (NWRI) to review the Phase II findings and to recommend future studies, further investigation into the role of the AES HBGS facility was one of the recommendations..

2.3 MBC AES HBGS INVESTIGATIONS

In response to the ocean postings and closures of 1999, AES HBGS commissioned MBC Applied Environmental Services (MBC) to conduct an in-plant investigation of bacterial concentrations at a variety of locations within and adjacent to the AES HBGS facility in the summer of 2001. Water quality samples were collected between May 30 and September 28, 2001. MBC identified high concentrations of indicator bacteria within the discharge vault at the AES HBGS. MBC identified the PCH/Newland Street storm drain (offsite) and the east and west retention basins (onsite) as the sources of the indicator bacteria in the discharge vault; however, MBC concluded that the AES HBGS facility was not responsible for bacterial exceedances reported at 9N on the Huntington State Beach because of initial dilution within the discharge vault and subsequently within the immediate vicinity of the ocean outfall (MBC, 2002). The RWQCB is currently revising the AES HBGS NPDES permit to include a bacterial discharge limitation (Theisen, 2002) for the AES HBGS.

2.3.1 GENERAL

MBC collected samples three times a week between 10:15 to 12:50 from May to September 2001. Samples were collected from the following 15 locations:

- Discharge vault;
- Discharge vault - bottom depth;
- Discharge vault - middle depth;
- Discharge vault - surface;
- East retention basin;
- East retention basin inflow;
- Intake forebay;
- PCH/Newland storm drain;
- Retention basin sump well;
- Units 1 and 2 sump;
- Units 3 and 4 sump;
- West retention basin;
- West retention basin inflow pipe;
- West retention basin outflow; and
- West retention basin well.

Each sample was analyzed for total coliform, *E. coli*, and Enterococcus using the Colilert (total coliform and *E. coli*) and Enterolert (Enterococcus) test methods, respectively. The two test methods have a lower detection limit of 10 MPN/100 ml and an upper limit of detection of 24,192 MPN/100 ml. Mean concentrations have been derived using the lower or upper limit as the actual value, which can lead to an underestimation of the mean bacterial concentrations. Circulating water flow through the plant was generally low in June and July (zero to three circulating pumps), but increased from early August through late September (four to seven circulating pumps). Intake forebay and discharge vault water temperatures were recorded along with tidal position, salinity, direction of cooling water flow within the AES HBGS, and the number of circulating pumps operating. During the summer of 2001, units 3 and 4 at the AES HBGS were retooled (rebuilt) to increase power production; as a consequence, there was a significant increase in the level of contractor and general on-site activity, including the use of portable restrooms.

2.3.2 DISCHARGE VAULT

A total of 58 total coliform samples were collected from the discharge vault (44 mid-depth samples, one bottom-depth sample, six at the surface, and seven at an unspecified depth) between May 30 and September 28, 2001. Total coliform concentrations ranged from below the detection limit (10 MPN/100 ml) to above the detection limit (greater than 24,192 MPN/100 ml). The majority of samples were between 10 and 1000 MPN/100 ml (39 samples). A total of 58 *E. coli* samples were collected from the discharge vault (44 mid-depth samples, one bottom-depth sample, six at the surface, and seven at an unspecified depth) between May 30 and September 28, 2001. *E. coli* concentrations ranged from below detection limits to 1,296 MPN/100 ml. The majority of samples were between 10 and 100 MPN/100 ml (53 samples). A total of 58 Enterococcus samples were collected from the discharge vault (44 mid-depth samples, one bottom-depth sample, six at the surface, and seven at an unspecified depth) between May 30 and September 28, 2001. Enterococcus concentrations ranged from below detection limits to 1,694 MPN/100 ml. The majority of samples were between 10 and 100 MPN/100 ml (52 samples).

2.3.3 INTAKE FOREBAY

A total of 51 total coliform samples were collected from the intake forebay between May 30 and September 28, 2001. Total coliform concentrations ranged from below detection limits to above detection limits. The majority of samples were between 10 and 100 MPN/100 ml (35 samples). A total of 51 *E. coli* samples were collected from the intake forebay between May 30 and September 28, 2001. *E. coli* concentrations ranged from below detection limits to 487 MPN/100 ml. The majority of samples were below detection limits (44 samples). A total of 51 Enterococcus samples were collected from the intake forebay between May 30 and September 28, 2001. Enterococcus concentrations ranged from below detection limits to 12,997 MPN/100 ml. The majority of samples were below detection limits (40 samples).

2.3.4 EAST RETENTION BASIN

A total of 13 total coliform samples were collected from the east retention basin between June 29 and August 1, 2001. Total coliform concentrations ranged from below detection limits to above detection limits. The majority of samples were near or above detection limits (seven samples). A total of 13 *E. coli* samples were collected from the east retention basin between May 30 and September 28, 2001. *E. coli* concentrations ranged from below detection limits to 464 MPN/100 ml. The majority of samples were between 50 and 400 MPN/100 ml (eight samples). A total of 13 Enterococcus samples were collected from the east retention basin between May 30 and

September 28, 2001. Enterococcus concentrations ranged from below detection limits to 41 MPN/100 ml. The majority of samples were below detection limits (nine samples).

2.3.5 WEST RETENTION BASIN

A total of 32 total coliform samples were collected from the west retention basin between June 11 and September 28 2001. Total coliform concentrations ranged from below detection limits to above detection limits. The majority of samples were above 8,000 MPN/100 ml (19 samples). A total of 32 total *E. coli* samples were collected from the west retention basin between June 11 and September 28, 2001. *E. coli* concentrations ranged from below detection limits to 9,804 MPN/100 ml. Nine samples were above 1,000 MPN/100 ml. A total of 32 Enterococcus samples were collected from the west retention basin between June 11 and September 28 2001. Enterococcus concentrations ranged from below detection limits to 609 MPN/100 ml. The majority of samples were below 100 MPN/100 ml (23 samples).

2.3.6 PCH/NEWLAND STORM DRAIN

The PCH/Newland storm drain represented an off-site sample location. Three total coliform samples were collected from the PCH/Newland storm drain on June 11, 13, and 15, 2002. Total coliform concentrations were above detection limits on the first two sample occasions and were 17,329 MPN/100 ml on June 15, 2001. Three *E. coli* samples were collected at the same time. *E. coli* concentrations were 1,793; 305; and 86 MPN/100 ml on June 11, 13, and 15, respectively. Three Enterococcus samples were collected at the same time. Enterococcus concentrations were 663, 134, and 10 MPN/100 ml on June 11, 13, and 15, respectively.

2.3.7 RETENTION BASIN SUMP WELL

The retention basin sump was sampled once on June 15, 2001. Total coliform, *E. coli*, and Enterococcus concentrations were 86, <10, and <10 MPN/100 ml, respectively.

2.3.8 UNITS 1 AND 2 SUMP, AND UNITS 3 AND 4 SUMP

Both sample locations were sampled on June 15, August 3, and August 8, 2001. Total coliform concentrations were 464; 1,354; and 4,611 MPN/100 ml; *E. coli* concentrations were 10, 158 and <10 MPN/100 ml; and Enterococcus concentrations were <10, 41, and 292 MPN/100 ml in Units 1 and 2 sump on the respective dates. In Units 3 and 4, total coliform concentrations were 31, 31, and 20 MPN/100 ml; *E. coli* concentrations were all below detection limits; and Enterococcus concentrations were 109, 10, and <10 MPN/100ml on the respective dates.

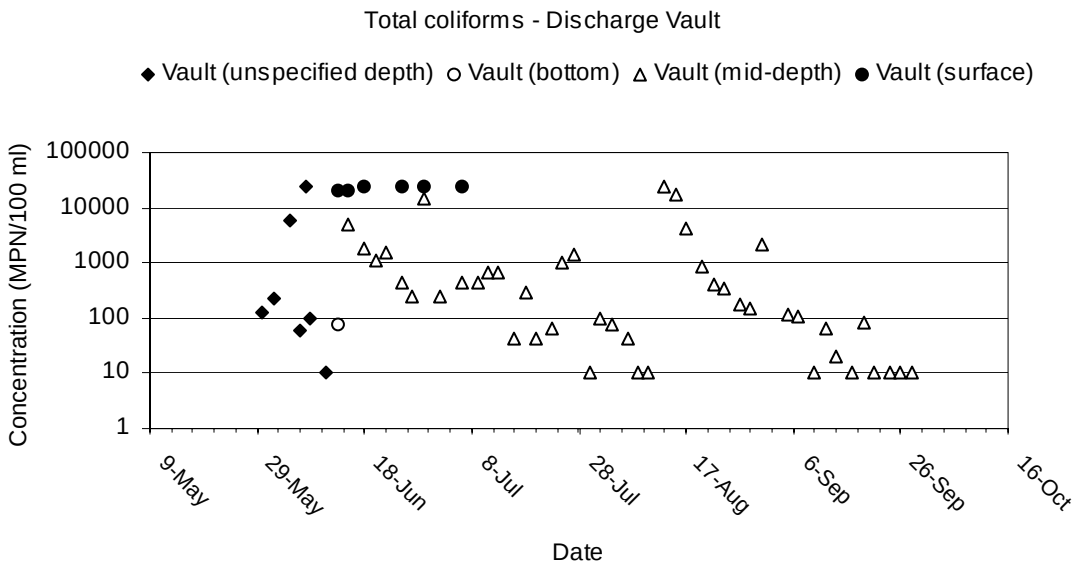
2.3.9 HEAT TREATMENT

Between July 1998 and September 2002, heat treatments were performed within the AES HBGS on the following dates: August 11, November 10, and December 24, 2001; February 15, April 2, June 15, August 4, September 15, and November 10, 2002 (Kunz, 2002). The heat treatment process of November 10, 2002, was particularly aggressive, with a maximum observed temperature of 47°C (approximately 120°F). The longevity of heat treatment processes can vary from six to twelve hours. No heat treatments coincided with sampling conducted by MBC in tidally influenced locations (intake forebay and discharge vault), although a heat treatment process occurred on August 11 2001 with subsequent samples on August 13 showing elevated levels of total coliform, *E. coli* and Enterococcus in the discharge vault and intake forebay. During this period, no bacterial exceedances were observed at ocean monitoring stations on Huntington State Beach. It is not clear whether this particular heat treatment process carried over to August 13, 2001 and whether this may have influenced bacterial concentrations. No statistically significant trend was observed between heat treatment and bacterial concentrations in tidally influenced sample locations immediately after the heat treatment process.

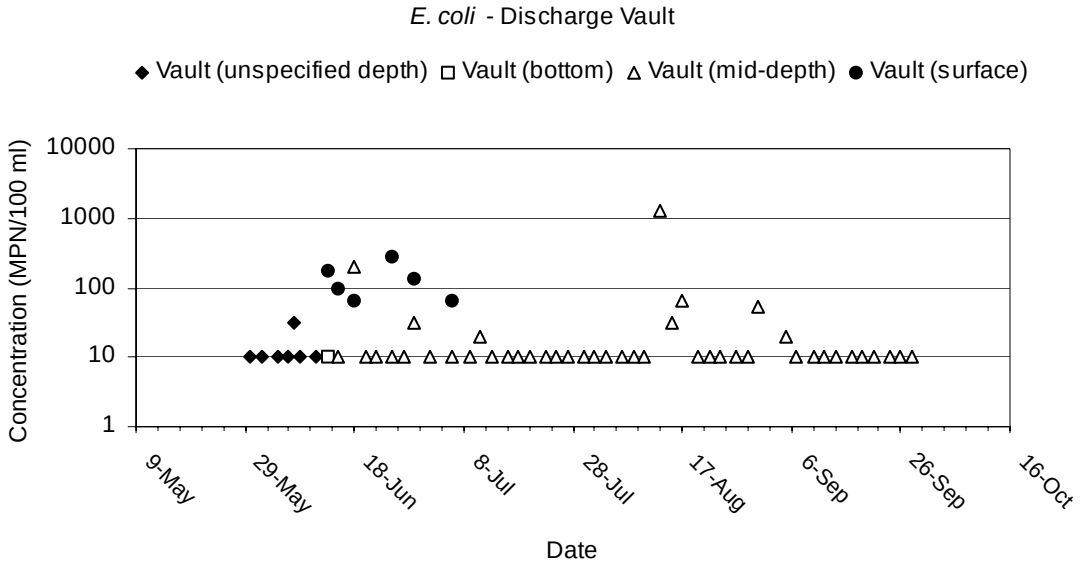
2.3.10 TIDALLY INFLUENCED BACTERIAL CONCENTRATIONS

Bacterial concentrations in tidally influenced sampling locations (intake forebay, discharge vault, and PCH/Newland Street storm drain) were highly variable. Generally, the intake forebay (total coliform mean: 1,537 MPN/100 ml; *E. coli* mean: 26 MPN/100 ml; and Enterococcus mean: 271 MPN/100 ml) were significantly lower than the discharge vault (total coliform mean: 4,267 MPN/100 ml; *E. coli* mean: 51 MPN/100 ml; and Enterococcus mean: 79 MPN/100 ml) and the PCH/Newland Street storm drain (total coliform mean: 21,904 MPN/100 ml; *E. coli* mean: 728 MPN/100 ml; and Enterococcus mean: 269 MPN/100 ml). Bacteria concentrations in the discharge vault appeared to be strongly influenced by fresh water inputs. Bacteria concentrations in the discharge vault decreased with depth and salinity (as shown in **Figure 2-1** through **2-3** below: -depth and salinity being positively related)

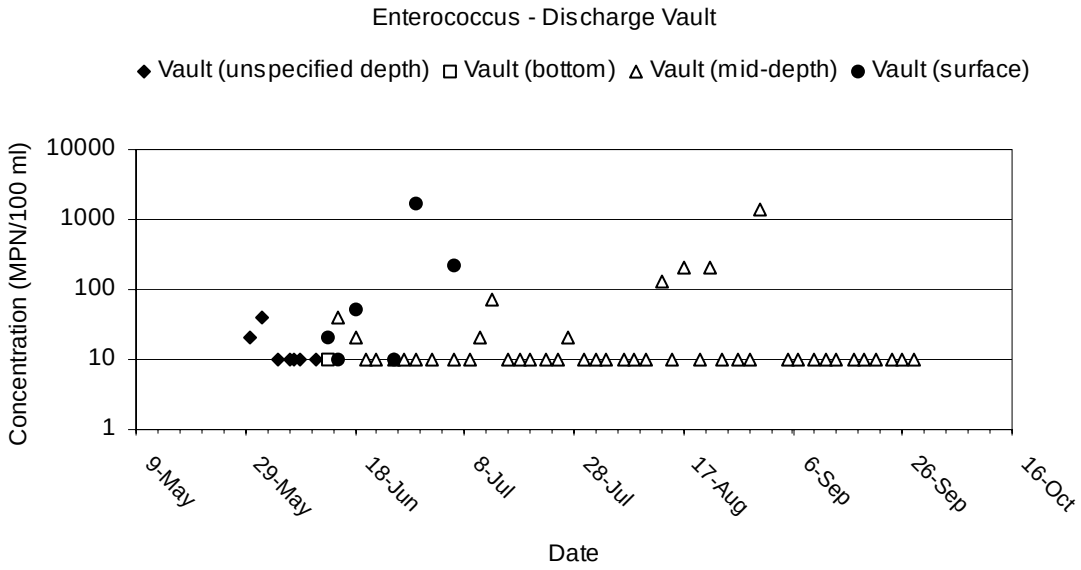
Total coliform concentrations in the discharge vault in the summer of 2001 (MBC, 2002).



***E. coli* concentrations in the discharge vault in the summer of 2001 (MBC, 2002)**



Enterococcus concentrations in the discharge vault in the summer of 2001 (MBC, 2002)



High concentrations of all indicator bacteria in the discharge vault coincided with high concentrations in the storm drain and the west retention basin, suggesting a role for both the storm drain and west retention basin in bacterial discharge from the AES HBGS. In-plant surface runoff contributes to the bacterial concentrations in the west retention basin, whilst a variety of sources (mobile home park, dog sanctuary, and urban runoff) potentially contribute to bacterial concentrations in the PCH/Newland Street storm drain. The maximum concentrations observed for Enterococcus (*E. coli* is not specifically referenced in AB 411 regulations, and many total coliform concentrations exceeded 24,192 MPN/100 ml) will be retrospectively applied as a discharge from the AES HBGS to the thermal plume model developed in this study.

2.3.11 NON-TIDALLY INFLUENCED BACTERIAL CONCENTRATIONS

Mean bacterial concentrations at non-tidally influenced locations within the AES HBGS are depicted in the table below. Bacterial concentrations in both the east and west retention basins were generally similar, whilst the sumps in both units were significantly lower. High total coliform concentrations were observed on one of three sample occasions for the west retention basin well, though the significance of this value is difficult to determine.

Mean Bacterial Concentrations in Non-tidally Influenced Sample Locations (MBC, 2002)

Sample Location	Total coliform (MPN/100 ml)	*E. coli* (MPN/100 ml)	Enterococcus (MPN/100 ml)
East retention basin	14,803	146	15
West retention basin	14,292	1,336	35
Retention basin sump well	86	<10	<10
Units 1 and 2 sump	2,143	59	114
Units 3 and 4 sump	27	10	43
West retention basin well	9,658	175	237

2.3.12 SOURCES OF IN-PLANT BACTERIAL CONTAMINATION

The high concentrations of bacteria in the retention basins may be attributable to anecdotal evidence at the time suggesting a direct input of fecal material into the two basins. If such an event did indeed occur, it would explain the high concentrations observed in not only the basins, but also in the receiving water in the discharge vault, and possibly may also have been responsible for high concentrations observed in the PCH/Newland Street storm drain (at high tide, water from the discharge vault may back up into the PCH/Newland storm drain).

Other potential sources of bacterial contamination within the plant suggested in the 2001 MBC study would include the relatively high density of pigeons roosting in the HGBS facility, cats observed on the facility, general urban runoff, and the possibility of an inappropriately plumbed restroom. Despite the chlorination of the west retention basin on August 8, 2001, elevated total coliform, *E. coli*, and Enterococcus concentrations persisted for the following six weeks.

2.3.13 RELATIONSHIP BETWEEN AES HBGS BACTERIAL CONCENTRATIONS AND OCEAN MONITORING DATA

Bacterial concentrations within the intake forebay are generally representative of surf zone bacterial concentrations collected at 9N (assuming samples were collected within the plant at a similar time to the OCSD sampling). The relationship between bacterial concentrations within the intake forebay and at 9N is shown in the **Table 2-2** (significant disparities between intake forebay and 9N bacterial concentrations are highlighted).

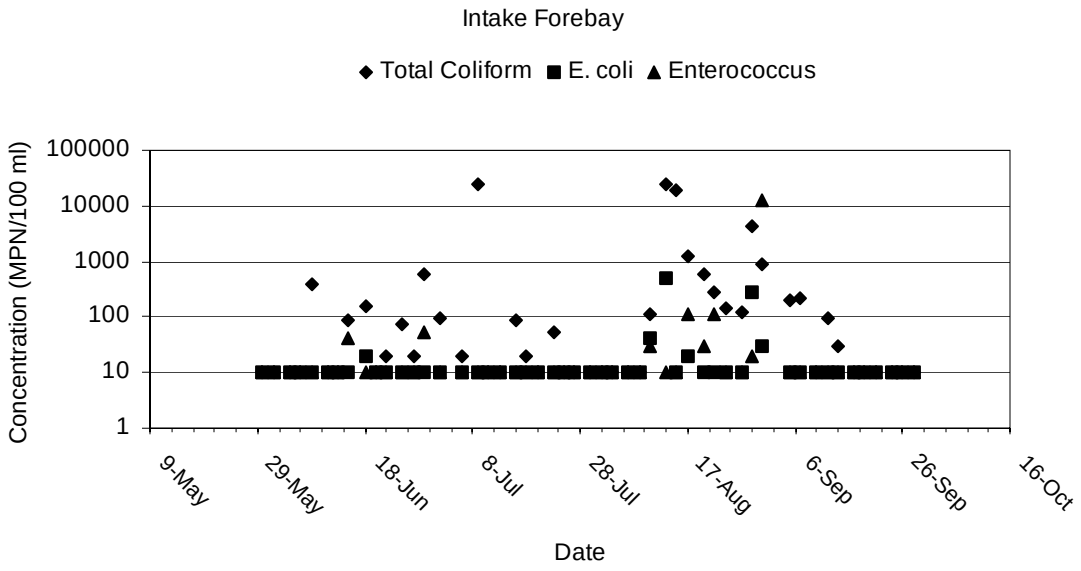
Bacterial concentrations at 9N and the Intake Forebay in 2001 (MBC, 2002).

Date (2001)	Sample Location	Total coliform (MPN/100ml)	*E. coli* / Fecal coliform (MPN/100ml)	Enterococcus (MPN/100ml)
May 30	AES HBGS	10	<10	<10
	9N	80	80	2
June 4	AES HBGS	<10	<10	<10
	9N	130	130	30
June 6	AES HBGS	10	<10	<10
	9N	300	300	18
June 11	AES HBGS	<10	<10	<10
	9N	20	20	6
June 13	AES HBGS	<10	<10	<10
	9N	20	<20	8
June 18	AES HBGS	161	20	<10
	9N	430	240	250
June 27	AES HBGS	20	<10	<10
	9N	<20	<20	<2
July 2	AES HBGS	98	<10	<10
	9N	20	20	4
July 11	AES HBGS	10	<10	<10
	9N	80	80	12
July 16	AES HBGS	85	<10	<10
	9N	<20	<20	<2
July 18	AES HBGS	20	<10	<10
	9N	250	140	140
July 23	AES HBGS	52	10	<10
	9N	40	20	60
July 25	AES HBGS	10	10	<10
	9N	30	30	20
July 30	AES HBGS	10	<10	<10
	9N	<20	<20	8
August 1	AES HBGS	<10	<10	<10
	9N	40	40	14
August 6	AES HBGS	10	<10	<10
	9N	<20	<20	<28
August 8	AES HBGS	10	<10	<10
	9N	110	70	40
August 13	AES HBGS	**>24,192**	**487**	10
	9N	**40**	**40**	20
August 15	AES HBGS	**19,863**	<10	<10
	9N	**40**	40	36
August 20	AES HBGS	**573**	<10	31
	9N	**>10**	20	<10
August 22	AES HBGS	**288**	<10	108
	9N	**<10**	<10	20
August 27	AES HBGS	120	<10	<10
	9N	110	110	202
August 29	AES HBGS	**4,352**	288	20
	9N	**110**	110	56
September 10	AES HBGS	<10	<10	<10
	9N	40	20	18
September 12	AES HBGS	97	<10	<10
	9N	<20	<20	30
September 17	AES HBGS	<10	<10	<10
	9N	10	20	<10

Between August 13, and August 29, 2001, there was a significant disparity between the bacterial concentrations observed at 9N and within the intake forebay; the reason is unknown. Bacterial concentrations were only tabulated when sampling events coincided between the intake forebay and 9N. Generally the concentrations of bacteria in the intake forebay were low and mirrored 9N **(Figure 2-4)**. For the month of August, water quality in the intake forebay deteriorated and

high concentrations were observed that did not correspond with surf zone water quality; the reason again is unknown.

Figure 2-4Bacterial Concentrations in the Intake Forebay in the Summer of 2001 (MBC, 2002)



2.4 BACTERIA IN COASTAL WATERS: A REVIEW

A review of bacteria in coastal waters has been included in Appendix A. The summary and conclusions of the review have been included below:

The survival of enteric (of the intestines) bacteria in the marine environment illustrates a set of attempts, many of them highly empirical in nature, to describe the actual fate of wastewater or enteric bacteria upon exposure to seawater. Only in recent years have these been expanded to include some molecular aspects of the studied phenomena. The purposes of the review have been to summarize and clarify knowledge and derive some insight into the variability of enteric bacterial populations in coastal areas affected by urbanization.

While it is generally accepted that when enteric bacteria are exposed to seawater there is a loss of colony-forming ability on solid media, there is controversy related to the physiological state of these non-culturable cells. Bogosian *et al.* (1996 and 1998) stated that viable but non culturable cells (VBNC), are either dead or of no significance as they cannot be practically resuscitated. In contrast, other evidence suggests that they are not only viable but that pathogenicity may be retained (Pommepuy *et al.*, 1996a, 1996b). The seawater-related VBNC debate is a part of the broader issue (Kell *et al.*, 1998; Barer and Harwood, 1999; Edwards, 2000)

of the true meaning of various viability criteria and of the molecular and biochemical mechanisms controlling the shift from colony forming to VBNC. These regulatory systems are only partially understood, and probably vary according to the stimuli imposed on the cells.

When enteric bacteria are exposed to seawater, they are simultaneously challenged by a combination of abiotic mortality factors; including the following:

2.4.1 LIGHT RADIATION AND ITS ASSOCIATED OXIDATIVE STRESS

Enteric bacterial survival in seawater is greatly affected by sunlight (ultraviolet particularly), light is considered to be the single most important factor in bacterial die-off in seawater. UV effects are restricted to shallow depths and has an important depth-dependant effect. Coliform bacteria are generally more susceptible to UV radiation than Enterococcus and fecal coliphages, and the effects of UV are more pronounced with increasing concentrations of dissolved oxygen, salinity increases and decreases in dissolved organic matter. UV radiation is believed to generate hydrogen peroxide and result in the oxidation of cellular lipid membranes.

2.4.2 HIGH SALINITY

Salinity or osmoregularity effects on bacteria can be enhanced by the presence of sunlight. On exposure to sea water bacteria can experience osmotic shock. Most bacterial cell walls are fairly resistant to moderate changes in osmotic pressure, however in hypertonic solutions (high salt or sugar concentrations) water diffuses out of the bacterial cell resulting in dehydration (plasmolysis); in hypotonic solutions (distilled water) the opposite effect occurs, and water diffuses into the cell and lysis occurs. Consequently salts and sugars are frequently used as food preservatives. In one experiment 92% of an *E. coli* population had not survived exposure to 100% seawater solution after 48 hours, 60% survived exposure to distilled water, and the optimum concentration was 25% seawater/ 75% distilled water (Carlucci and Pramer, 1960b).

2.4.3 HIGH PH

Seawater pH typically ranges from 7.5 to 8.5 and is influenced by temperature, pressure and microorganisms. Enteric microorganisms are adapted to the moderately acidic mammalian intestines of approximately pH 5.0, higher pH is increasingly deleterious to the survival of enteric bacteria.

2.4.4 LOW TEMPERATURE

Enteric bacteria have an optimal temperature growth range around 37°C, optimal survival does not necessarily coincide with optimal growth temperatures. Significantly greater survival at lower temperatures has been demonstrated, growth is none the less significantly reduced in anything but a narrow temperature range around the optimal growth temperature.

2.4.5 LIMITED NUTRIENT AVAILABILITY

Limited nutrient availability in seawater in conjunction with other factors can result in inactivation of membrane transport mechanisms and pronounced reduction in energy charge. Outside of contaminated coastal zones, bacteria essentially have to deal with starvation conditions.

2.4.6 POSSIBLE PREDATION AND COMPETITION

Among possible biotic factors that may play a role in determining survival, predation by protozoa was the only one shown to be potentially significant. The contribution of other effectors, such as bacterial predation, viral infections, or unexplained antibiotic-like effects, was either insignificant or not demonstrated under natural conditions.

2.4.7 SUMMARY

A recurring factor apparent from many studies is that in addition to actual seawater incubation conditions, previous growth history has a major influence on subsequent survival in the marine environment, suggesting significant adaptation potential. Such potential may play a role in recurring seawater contamination from stagnant waters or other coastal sources.

Many of the studies considered were motivated by the desire to protect near shore areas from fecal pollution and its ultimate effects on human health. In the design of marine sewage effluent outlets, bacterial die-off constants are often used (mostly *E. coli*). Such values may be based on both laboratory and field experiments, and it is of importance to recognize the limitation of both approaches. The significance of many of the field experiments is often site-specific, and tend to ignore previous growth history of the monitored strains. Thus, mathematical models based on such results cannot replace the need for routine bacterial pollution monitoring. Conversely, laboratory experiments can be accurately designed to test the effects of specific parameters, but cannot simulate or imitate the complexity of the real marine environment. As such, additional information about seawater currents, temperature, and turbidity may be needed to complement indicator count measurements.

Basic molecular studies may be able to markedly affect wastewater pollution monitoring practices through detection methodologies of either pathogenic or indicator microorganisms. Present approaches, based upon a few simple though highly informative indicator assays, may be replaced by several proposed molecular approaches that allow fast and sensitive detection of bacteria, viruses, and other organisms. Such methods may lead to a more accurate and rapid wastewater pollution detection, including bacteria in the VBNC state. These methods will provide a broader understanding of the survival of these organisms, thus meeting the challenge of future health management of marine waters.

3 METHODOLOGY

The scope of work included collection of water quality data within and adjacent to the AES HBGS and in the near shore ocean environment off Huntington State and City Beaches, thereby establishing a baseline of water quality. The water quality data was used to assess the four possible mechanisms (**Section 1.3**), that may be acting independently or collectively to impact the near shore environment of Huntington State and City Beaches.

By assessing the four potential mechanisms, the investigation has answered the two key questions identified by the Technical Advisory Group (TAG):

- Is the AES HBGS discharging, transporting, and/or pushing contaminated water to shore near to 9N?
- Are land-based sources (i.e. the intake from the near shore ocean environment, or the storm drains) conveying bacteria-contaminated urban runoff into the AES HBGS's discharge?

The following sections consist of an analysis of the area within and adjacent to the AES HBGS, and an analysis of the near shore ocean environment.

3.1 HUNTINGTON BEACH GENERATING STATION SAMPLE LOCATIONS

After discussions with the CEC, AES HBGS representatives, and the TAG, and a site walk on July 12, 2002, the following seven sampling locations were selected with an additional opposite flow sample location during heat treatments or cleaning, with three sets of samples to be taken at different depths at one of the locations:

- Boiler fireside wash (BFW);
- Boiler fireside wash sump (BSW);
- Storm water sump (SWS);
- General purpose retention basin (GP);
- Blackford's Ditch running alongside Newland Street (BD);
- The intake vault (IV);
- The discharge vault at the surface level (DV-0), DV-10, and DV-30; and
- Opposite Flow (OF), a sample location used during heat treatment periods.

The sampling point locations are within the areas shown on **Plate A-1**. Samples were collected in the sequence listed above from identical points at each location at the same time each day.

Sampling started each day at 06:00 to ensure that sampling occurred between 08:00 and 10:00 at the intake and discharge vaults in coordination with the County of Orange and the OCSD. Sample labels included not only the sample location but also the date collected, thereby becoming a unique identifier.

As part of this investigation, ammonia sensors were to be deployed on moorings and in the intake vault, however technical limitations of the equipment prevented the successful deployment of the ammonia sensors, consequently ammonia samples were collected from four locations within the AES HBGS facility (discharge vault at zero-feet, ten-feet, and 30-feet depths, and the intake vault).

3.1.1 THE BOILER FIRESIDE WASH BETWEEN UNITS ONE AND TWO

The BFW received wash runoff from around the base of units one and two, and from the wastewater drains for units one and two. A large population of birds was resident in the two units and consequently contributed a significant amount of fecal material to the floor of units one and two. The effect of bird feces on water quality in the plant was assessed by taking water samples from the BFW. The boiler fireside wash was located directly underneath units one and two, and field personnel working there required the use of earplugs for hearing protection. Water samples were collected from the floor drain located between the two units.

3.1.2 THE BOILER SUMP WASH

The BSW received water from all the floor drains in the boiler fireside wash area under units one and two, as well as wastewater from the shop, warehouse, unit one and two auxiliary bay drains, the peaker sump discharge, and flood drain. The BSW was located approximately 50 feet from the base of the smokestack, serving units one and two. The boiler wash sump contained approximately six inches of oil on the surface of the water; consequently, only microbial water quality parameters were assessed at this location.

3.1.3 THE GENERAL PURPOSE RETENTION BASIN

The GP was used as a bulk water storage facility and has heavy growth of macroalgae and, potentially, microorganisms. The GP drained into the discharge vault, and the flow rate was monitored. The rate of pumping was recorded at the same time as samples were collected. The GP was located at the southern boundary of the site, adjacent to the wildlife sanctuary alongside the PCH. The samples were collected from the open water adjacent to the discharge point from the GP. Although a vertical sampling pipe was available, the absence of UV light

may have resulted in the sample being unrepresentative of the actual bacterial population in the GP; consequently, samples were collected approximately one foot from the edge of the water in the basin, in a line between the sampling pipe and the depth measurement pipe. Samples were collected at mid-depth (approximately one foot deep) at this location.

3.1.4 BLACKFORD'S DITCH

BD running alongside Newland Street from Edison Avenue to the main gate of the AES HBGS site was a potential source of microorganisms. Approximately 100 feet of the ditch had water present to a depth of two feet. During the site visit on July 12, 2002, significant discoloration of the water adjacent to the entrance to the main gate was observed; the water appeared similar (in color) to automotive antifreeze or biocide used within the AES HBGS. The ditch was hydraulically connected to the discharge vault and, at high tides; ocean water from the discharge vault could have been forced back up into the ditch. In addition to the daily sampling from one location in the creek, a thorough investigation was conducted on the ditch with samples taken from several locations and analysis conducted for water quality parameters. Samples were also collected for microbial source tracking and identification. The thorough investigations were conducted on three separate occasions (August 13, 2002; September 9, 2002; and November 26, 2002); several grab samples were taken and analyzed specifically for the presence of components of the biocide used within the AES HBGS.

3.1.5 THE INTAKE VAULT

IV received incoming ocean water from the intake pipe and was located approximately 30 feet behind the GP and in front of the sweet water tanks adjacent to units three and four. Samples were collected from just below the water surface using a sampling container attached to a pole so that samples could be collected from outside the safety railing surrounding the IV.

3.1.6 THE DISCHARGE VAULT, AT THE SURFACE AND APPROXIMATELY TEN- AND 30-FEET DEPTH

The discharge vault received water from the GP, the storm drain sump (Site storm drains), BD and discharge cooling water. The discharge vault was located behind the GP and was approximately 50 feet west of the intake vault. Water quality samples were collected at three depths within the discharge vault; surface level, DV-10 and DV-30.

3.1.6.1 Discharge Vault-0

The depth below ground surface for the surface water sample varied consistently with tidal height, and was therefore reported as surface water level or DV-0.

3.1.6.2 Discharge Vault-10

On the first day of sampling, when attempting to achieve a depth below surface water level of ten feet, the sampling equipment was dragged diagonally under the water into the discharge pipe. Because the depth of the discharge pipe from the surface water level would vary consistently with tidal height throughout the course of the investigation, the entry point for the discharge pipe from the discharge vault was selected as the DV-10 sample point. Thereby representing a consistent sampling location of water being discharged to the ocean. The discharge pipe entrance lies at approximately 12 feet below ground surface and the DV-10 sample point actually represents a sample depth of 13 feet below ground surface. During periodic spring low tides sample points DV-0 and DV-10 could be vertically separated by three to four feet, during a standard tidal cycle, vertical separation between these two sample points would be between five to eleven feet.

3.1.6.3 Discharge Vault-30

The sample collected at 30-foot depth below ground surface (DV-30) was collected using a small 12-volt electric submersible pump lowered to the bottom of the discharge vault inside a two-inch stainless steel well screen. The well screen prevented the submersible pump from being drawn into the discharge pipe. During each day of sampling, a profile of the salinity in the discharge vault was generated using water samples collected from the water surface to the bottom of the discharge vault at one-foot intervals. The salinity of each water sample was measured in the field using a Horiba U-10 portable water quality meter.

3.1.6.4 Opposite Flow

During heat treatment processes, water is recirculated through the condensers in the AES HBGS to reduce and remove biofouling. Samples were collected at an alternate location during heat treatment processes, as the discharge vault sample locations would not be indicative of water discharged to the ocean during this time. Samples were collected from a location between the discharge vault and the intake vault through wooden slats; samples were identified as Opposite Flow (OF).

3.1.7 ADDITIONAL STORM DRAINS

Two additional storm drains are located in the vicinity of the AES HBGS and may have been a potential source of microorganisms if they were found to drain into the discharge vault. The first storm drain was located on the west side of Newland Street opposite the AES HBGS, and ran parallel to the PCH. A second storm drain was located underneath the wildlife sanctuary, south of the AES HBGS. Microbial samples of these storm drains were collected during a period of rainfall (November 9, 2002) to determine the water quality and to assess the connectivity of the ocean, storm drains, and the AES HBGS discharge vault.

3.2 ANALYTICAL PARAMETERS

3.2.1 LABORATORY ANALYTICAL PARAMETERS

Water samples were analyzed by Sierra Analytical Labs of Laguna Hills, California. Sierra Analytical is a State-of-California-approved environmental laboratory, certified by the Department of Health Services under the Environmental Laboratory Accreditation Program (ELAP) [Certificate Number 2320]. Each water sample was analyzed for the following microbiological constituents, in accordance with the associated analytical methods, techniques and hold times:

Total coliform

Standard total coliform test (Membrane filter procedure)	Standard Method 9222 B

Fecal coliform

Delayed-incubation fecal coliform test	Standard Method 9222 D

Enterococcus

Enterococcus (Multiple-tube) test	Standard Method 9230 C

In addition, three samples were collected daily from the intake vault and the discharge vault at the water surface and ten-feet depth. These samples were analyzed for the following:

Ammonia	EPA Method 350.1
Salinity	Standard Method 2520

Initially, Sierra Analytical analyzed salinity samples; however, after September 23, 2002, the laboratory selected for salinity samples was changed to Scripps Institute of Oceanography.

3.2.2 FIELD ANALYTICAL PARAMETERS

Water samples collected at the AES HBGS and at the three storm drains in the vicinity of the AES HBGS were analyzed in the field for the following parameters using a Horiba U-10 portable water quality meter:

- pH;
- Temperature;
- Conductivity;
- Salinity;
- Turbidity; and
- Dissolved oxygen (where appropriate).

The field equipment was calibrated each day to ensure proper functioning prior to analysis of the field samples.

In addition to using the Horiba for physical parameters, SeaBird MicroCat SBE 37-SM recorders (SB-MC) were deployed at the AES HBGS for three months to analyze the temperature and salinity (conductivity) at a higher frequency and accuracy. The SB-MC recorders were deployed in the intake vault and in the discharge vault at the surface and at 10 feet below the surface. After one month of operation, the SB-MC located at ten-feet depth in the discharge vault was lowered to 30 feet below the surface to provide more information of the salinity and temperature gradients within the discharge vault. Deployment was limited to the three SB-MC recorders due to the intermittent and variable volume of water located at the other locations. The SB-MC has a high accuracy conductivity and temperature recorder with internal battery and memory. The SB-MC memory can store in excess of 185,000 samples at a resolution of 0.0001 mS/cm for conductivity and 0.0001°C for temperature.

3.2.3 FIELD WORK

All water samples were collected in accordance with Standard Method 9060 A and preserved and stored in accordance with Standard Method 9060 B. The generalized sample collection procedure for field parameters and microbial analysis was as follows:

- Sample bottles were held near their base and submerged neck downward below the surface of the water (where possible);
- The neck and mouth of the sample bottle were pointed slightly upward towards the direction of the current, if a current was present;

- Care was taken to avoid contact with the edge of the pond, channel, or other structures that would support the growth of a microbial biofilm;
- A small volume of flowing water (approximately 200 ml) was collected in the Horiba U-10 water sample container for analysis of field parameters in the discharge vault. For all other locations, the Horiba was lowered into position approximately four inches beneath the surface of the water; and
- The sample containers for this type of sample consist of three 100 ml polyethylene bottles containing sodium thiosulfate as a preservative.

The following procedures were followed to avoid cross-contamination or sample bias:

- All personnel, upon arrival at the site, thoroughly washed their hands with anti-microbial soap prior to beginning water sampling;
- Medical examination gloves were worn by all sampling personnel when handling sampling containers. A new set of gloves were worn for each new sampling location and were only put on immediately prior to sampling;
- Each sample container was pre-labeled prior to the collection of water samples;
- Lids were kept on all sample containers until the sample was to be collected. At that time, the sample container (with lid still on) was brought near the sampling location. The lid was unscrewed, and at no time did the mouth or rim of the sample container come into contact with the sampler, sampling equipment, or any other non-sterile material. Once an adequate volume of water was obtained, the container was removed from the surface of the water and the lid replaced and firmly tightened;
- Full sample containers were placed into new plastic sealable bags. One bag was used per sample set;
- Each plastic sealable bag used was labeled on the outside using a permanent pen;
- Each sample bag was immediately placed inside a cooler with ice packs to maintain an internal temperature of approximately 4°C. The cooler was kept closed and away from direct sunlight, minimizing the exposure of the samples to UV radiation and helping to maintain a near-constant, low temperature inside the cooler;
- Chain-of-custody procedures were followed for all samples collected;
- Sampling commenced every day at 06:00 in the morning and was completed by 11:30; and
- Microbiological samples were collected by a representative from Sierra Analytical to ensure the hold time of six hours for fecal coliform analysis was not exceeded.

3.2.4 SAMPLING SCHEDULE

A three-month onsite AES HBGS sampling schedule was initiated on July 16, 2002. Between August 15 and 29, 2002, an intensive two-week sampling period occurred to coincide with spring tides.

3.2.4.1 Three-Month Sampling Period

During this sampling period, the seven sample location sites within the AES HBGS vicinity were sampled on a daily basis. The following quality assurance/quality control (QA/QC) samples were collected daily: one field blank, one equipment blank, and one duplicate sample. Additionally, three MicroCat recorders were installed at the AES HBGS during this time. In the near shore ocean environment, mooring sensors were deployed in August.

3.2.4.2 Two-Week Intensive Sampling Period

During the two-week intensive sampling period, the timing of sampling at the AES HBGS coincided with near shore ocean investigations and was coordinated with the OCSD and the County of Orange beach sampling activities. For two weeks, the four tidally influenced sample locations (the intake vault, two discharge vault locations, and Blackford's Ditch) in the vicinity of the AES HBGS were sampled every three hours.

3.2.5 DOCUMENTATION

3.2.5.1 Field Notes

Field notes for the sampling events consisted of the following:

- A Daily Field Report Form for each field day **(Appendix B).** The form notes the project name, project number, date, number of pages, Komex personnel present on site, a general work description, and detailed individual entries for each work activity carried out that day, general Site observations made during sample collection activities at each location, and documentation from AES HBGS personnel, including the temperatures of intake and discharge vaults, the number of operational units, the use of chemical treatment, and the number of operational pumps;
- A Field Sampling Form for each field day **(Appendix C).** The form notes project name, project number, date, time, Komex employee names, number of pages, and detailed individual entries for each field sample analyzed. The detailed information includes sample

collection time, sample location, sample depths, sample analysis, and general remarks on the visual appearance of the sample; and

- Photographic documentation of general Site conditions based upon the observations at the time and photographs of water collection sampling activities were taken **(Appendix D).**

3.2.5.2 Chain-of-Custody Procedures

The methodology described within this section is in general accordance with the procedures described in American Society of Testing and Materials (ASTM) Standard D4840-88 (1993): Practice for Sampling Chain-of-Custody Procedures.

After sample collection and labeling, samples were maintained under chain-of-custody-form (COCF) procedures. The procedures document the transfer of custody of samples from the field to the laboratory. Each sample sent to the laboratory for analysis was recorded on a COCF, which included instructions to the laboratory for analytical services. Information contained on the triplicate COCF included the following:

- Project number;
- Signature of sampler;
- Date and time sampled;
- Sample I.D.;
- Number of sample containers;
- Sample matrix (water);
- Analyses required;
- Remarks, including preservatives, special conditions, or specific quality control measures;
- Turnaround time and person to receive laboratory report;
- Method of shipment to the laboratory;
- Release signature of sampler, and signatures of all people assuming custody; and
- Condition of samples when received by laboratory, including temperature.

The field sampler signed the COCF and recorded the time and date at the time of transfer to the laboratory or to an intermediate person. A set of signatures was required for each relinquished/reserved transfer, including transfer within Komex. The original imprint of the COCF accompanied the sample containers. A duplicate copy was placed in the project file (**Appendix E**).

3.2.5.3 Decontamination Procedure

All items used for sample collection were cleaned prior to and between sampling locations using the Standard Practice for Decontamination of Field Equipment Used at Non-radioactive

Waste Sites as described in ASTM Method D5088. In brief, items were cleaned using a brush and Alconox solution, followed by two deionized water rinses.

3.3 DYE TRACER STUDY OF THE SANITARY SEWERS

A dye tracer test was performed on September 16, 2002, and September 17, 2002, to evaluate whether the sanitary sewers within the AES HBGS were connected to, or leaking into, the discharge vault. A liquid tracer dye (Rhodamine WT) was added to the sewer system via the onsite washroom facilities. The six washroom facilities at the site are located in the following areas:

- Front administration building;
- Boilers 1 and 2 control room;
- Boilers 3 and 4 control room;
- Shop area;
- Peaker control room; and
- Edison Pipeline Storage Facility.

A Seapoint Rhodamine fluorometer was installed in the discharge vault so that the dye sensor was submerged at a depth below the crown of the discharge pipeline. The fluorometer was connected to an analog to digital (A to D) signal converter that transformed the analog voltage signal from the fluorometer into a digital signal that could be interpreted by a laptop computer. The A to D signal converter was connected to a laptop computer where the fluorometer software continuously logged the signal from the fluorometer. The fluorometer was set at a sampling interval of one Hertz (Hz). Data was logged at one-second intervals for 24 hours commencing at 10:00 on September 16, 2002. Approximately 86,400 data points were collected during the sampling period. The fluorometer sensitivity was set to record a voltage range of zero volts to five volts (representing a Rhodamine concentration of zero parts per billion [ppb] to five ppb, respectively).

Additionally, a video camera was installed to continuously record the water surface in the discharge vault during the 24-hour sampling period. A video light was also installed in the discharge vault to ensure that adequate light was available for good color resolution throughout the course of the study. Video was recorded onto blank Hi-8 cassettes. Every two hours, monitoring personnel put a new blank tape in the video camera so that a near-continuous video record of the water surface in the discharge vault was obtained.

Sampling personnel who were collecting water samples from the seven sampling locations on and around the site were instructed to report any observation of red dye at any of the seven sampling locations during the 24-hour study period.

Approximately 0.5 Liters (L) of concentrated Rhodamine WT fluorescent dye were simultaneously added to toilets at each of the six washroom locations on site (three liters of Rhodamine WT in total) at 10:45 on September 16, 2002. Each toilet was then flushed approximately 15 times to ensure the dye was carried into the sanitary sewer system.

Using data collected from the Rhodamine fluorometer, observations from the video camera records, and observations by sampling personnel over the 24-hour study period, a determination was made as to whether or not the sanitary sewers within the AES HBGS were connected to, or leaking into, the discharge vault.

3.4 NEARSHORE OCEAN ENVIRONMENT

The near shore ocean environment of the AES HBGS was assessed to determine whether the AES HBGS was drawing in and/or discharging microbiologically contaminated water. The thermal discharge plume was evaluated during an offshore dye study and modeled to determine how the plume diluted with ambient seawater and how it behaved in the marine environment. This included measurements of dye concentrations and predictions of likely shoreline contact points and the dilution values at those locations as they relate to recreational ocean water quality parameters. The goals of the offshore dye study and thermal plume model were to evaluate the following questions:

- Is the AES HBGS discharge of microbiologically contaminated water reaching the shoreline? If so, where is that occurring? and;
- Is the AES HBGS discharge causing or contributing to recreational water quality closures in the area?

3.4.1 INTAKE AND OUTFALL ASSESSMENT

Moored time-series of the temperature, conductivity, and nutrients were collected near the intake and discharge pipes of the AES HBGS. The purpose of these measurements is to examine the near shore variability of the temperature and salinity (T/S) field with the intent of resolving potential source water entering the cooling system of the generating system.

Each of the moorings had a T/S sensor about one meter above the bottom and one T/S sensor about one meter beneath the surface. The two most likely sources of potentially contaminated,

low salinity water are; landward sources, including the various river systems; and the OCSD outfall located offshore from AES HBGS. Resolution of the T/S field is important, but T/S alone does not provide unambiguous identification of effluent plume from offshore. Like salinity, elevated ammonia could also come from landward sources, but if the T/S characteristics of the water are typical of sewage effluent plume (T<13°C, S<33.5 psu), and ammonia is elevated, there is reasonable confidence in identifying the source of the water as the sewage outfall. T/S was sampled at two- to three-minute intervals to provide high frequency resolution that would approximate the frequency of internal waves with a Brunt-Vaisala frequency of 20 to 30 cycles per hour. Nutrient sensors were to be deployed at the intake mooring and in the intake vault to measure ammonia concentrations. The nutrient sensor deployed in the intake vault did not perform satisfactorily and consequently neither sensors were deployed at their planned locations. Water samples were collected from the intake vault, discharge vault and other sampling locations and anlyzed for ammonia by a certified laboratory.

In addition to the near-bottom and near-surface sensors, small temperature sensors were placed at one-meter intervals on the mooring to resolve the details of the temperature structure. This was important for understanding how internal waves interact with the intake and discharge and for examining the role of heating from the AES HBGS discharge in the upper two or three meters beneath the surface.

3.4.2 OFFSHORE DYE STUDY

Historical studies have demonstrated that the thermal plume from the AES HBGS reaches the ocean surface (by design) and that a large portion of these flows are then adverted onto the beach by the waves in the high-energy environment of the surf zone.

The purpose of the offshore dye study was to attempt to quantify and qualify the behavior of the thermal plume in the near shore environment including likely shoreline contact locations and the dilution values at those locations as they relate to recreational ocean water quality parameters.

The offshore dye study was conducted concurrently with the intensive in-plant sampling program so that both data sets could be compared and examined for statistical correlations.

3.4.2.1 Sampling Program

The offshore dye study was conducted between 07:00 on Tuesday, August 20 and 15:30 on Wednesday, August 21, 2002. The generalized procedure for the dye study was as follows (specifics are detailed after this summary):

- Rhodamine WT dye was injected into the discharge pipe of the AES HBGS outfall at a known concentration. The dye was injected as a series of five pulses throughout the day on August 20, 2002;
- During and after the dye injections, dye concentrations were measured using several methods, including a boat with a towed fluorometer, a helicopter with a high-resolution digital camera taking time-lapse photographs of the dye field, and personnel stationed along the shoreline at Stations 6N, 7.5N, 9N, 10.5N, 12N, 13.5N, and 15N;
- The boat (with towed fluorometer) measured the Rhodamine WT concentrations in the ocean as a function of position and time. Position was logged using a differential global positioning system (DGPS). Additional physical oceanographic parameters including conductivity (salinity), temperature, depth (pressure), pH, and DO were measured using a multi-parameter sonde. Water samples were also collected for analysis of bacteriological indicators (total coliform, fecal coliform, and Enterococcus) and microbial source tracking analyses;
- The helicopter with high-resolution digital camera was on station taking time-lapse photographs of the dye field;
- Personnel were stationed along the shoreline at Stations 6N, 7.5N, 9N, 10.5N, 12N, 13.5N and 15N. Personnel collected water samples (for laboratory analysis of dye) at intervals of 15 minutes from 09:30 until 20:00 on Tuesday, August 20, then at 30-minute intervals after that until the completion of the dye study at 15:30 on Wednesday, August 21, 2002; and
- In conjunction with the intensive in-plant sampling program that was occurring during the offshore dye study, personnel along the shoreline also collected water samples for microbial analysis at 6N, 9N, and 12N at intervals of three hours from 09:30 on Tuesday, August 20 until 12:30 on Wednesday, August 21, 2002. These samples were analyzed for total coliform, fecal coliform, and Enterococcus. Water samples were also collected for microbial source tracking analyses.

Five releases of Rhodamine WT dye (20% w/v concentration) occurred on August 20, 2002. Dye was injected directly into the discharge pipe of the AES HBGS outfall at a known rate. Table 3-1 summarizes the details of the dye injection program, including dye injection number, date, start time, end time, dye injection rate, and outfall dye concentration.

The offshore portion of the dye study was conducted from a 25-foot Boston Whaler. Equipment that was mounted on this vessel for the dye study included the following:

- Garmin GPSMAP 188 Sounder, Wide Area Augmentation System (WAAS) Enabled Global Positioning System (GPS) (with data acquisition and plotting software MapSource BlueChart version 4.01) linked to a laptop computer;
- Seapoint Rhodamine Fluorometer (with data acquisition software Data Translation version 2.2.0.30) linked to a laptop computer; and
- YSI 6600 Multi-Parameter Sonde (with data acquisition software Ecowatch version 3.13.10).

Table 3-1 Details of the Dye Injection Program

Dye Injection (#)	Date	Start Time (hh:mm)	End Time (hh:mm)	Injection Rate (ml/min)	Outfall Concentration (µg/L)
1	20-Aug-02	10:35	10:45	260	62
2	20-Aug-02	12:01	12:31	300	72
3	20-Aug-02	13:30	14:00	300	72
4	20-Aug-02	15:30	15:56	300	72
5	20-Aug-02	17:30	17:32	300	72
5a	20-Aug-02	17:40	17:45	200	48

Positioning of the vessel during the offshore dye study was accomplished using a Garmin GPSMAP 188 Sounder, WAAS-Enabled GPS linked to a laptop computer. In WAAS-Enabled mode, positional accuracy was to within three meters.

Rhodamine WT concentrations in the ocean were measured using a Seapoint Rhodamine Fluorometer. The fluorometer was deployed from a winch on the starboard side of the vessel. The fluorometer was set near the ocean surface at an approximate depth of 0.2 to 0.5 meters. Horizontal transects were conducted along the surf zone from the AES HBGS outfall to the Huntington Beach pier. Cross-sectional transects were conducted through the dye plumes after each of the five dye releases. Data were continuously collected while underway during the survey days. The fluorometer measured an analog voltage signal, which was converted into a digital signal using an A to D signal converter. The data acquisition software then logged this digital signal. Rhodamine concentrations in the following sections are expressed in micrograms per liter (µg/L), which are equivalent to parts per billion (ppb).

Physical oceanographic parameters were measured using an YSI 6600 Multi-Parameter Sonde. The YSI 6600 was deployed from the port side of the vessel. The YSI was attached to a winch system allowing it to be raised and lowered through the water column as required (Deployment rate 0.06 m/s [average]; sampling rate at 2 s intervals and response time of <1 s [increased accuracy by averaging every 4 s]; YSI 6600 conductivity resolution at 0.1 mS/cm and accuracy of ±5% + 1uS compared to the SeaBird conductivity resolution of 0.04 mS/cm and accuracy of ±0.003 S/m).

Table 3-2 Offshore Dye Monitoring Methodology

DAY	TOTAL HOURS	STATION	ONBOARD ACTIVITIES
20 August 2002	Ten	The equipment was deployed at the AES HBGS outfall, with continuous monitoring along the surf zone between the Huntington Beach Pier and the AES HBGS outfall.	Continuous DGPS positioning and logging; Background Rhodamine WT measurements; Background water quality measurements; Continuous ocean surface Rhodamine WT measurements with discrete sampling; Conductivity, temperature, depth, pH, and DO measurements; and Vertical profiles of conductivity, temperature, depth, pH, and DO at discrete stations.
21 August 2002	Four	The equipment was deployed at the Santa Ana River, with continuous surface monitoring along the surf zone to the Huntington Beach Pier.	Continuous DGPS positioning and logging; Background Rhodamine WT measurements; Background water quality measurements; Continuous ocean surface Rhodamine WT measurements with discrete sampling; Conductivity, temperature, depth, pH, and DO measurements; Vertical profiles of conductivity, temperature, depth, pH, and DO at discrete stations; Discrete water sampling (using a Niskin bottle) for bacteria (at 6 stations and two depths [ocean surface and near-bottom]); and Discrete water sampling (using a Niskin bottle) for microbial source tracking analysis (at two depths [ocean surface and near-bottom]).

Data were stored in the YSI memory until they could be downloaded to a laptop computer. The YSI 6600 logged data that included the following:

- Conductivity (salinity);
- Temperature;
- Depth (pressure);
- pH; and
- Dissolved oxygen.

Table A-1 summarizes the details of the physical oceanographic parameter vertical profile locations including cast number, latitude, longitude, and bottom depth.

The sonde calculates salinity automatically using the conductivity and temperature readings in accordance with algorithms found in *Standard Methods for the Examination of Water and Wastewater* (1998). The Practical Salinity Scale is used for unit-less values, since the measurements are carried out with reference to the conductivity standard of seawater at 15°C.

To measure temperature, the sonde utilizes a thermistor of sintered metallic oxide that changes predictably in resistance with temperature variation. The algorithm for conversion of resistance to temperature is built into the sonde software, and accurate temperature readings in °C is provided automatically. No calibration or maintenance of the temperature sensor is required.

For measuring pH, the pH probe is a combination electrode consisting of a proton-selective glass reservoir filled with buffer at approximately pH 7 and a silver / silver chloride (Ag/AgCl) reference electrode that utilizes an electrolyte that is gelled. A silver wire coated with Ag/AgCl is immersed in the buffer reservoir. Protons (hydrogen ions [H^+]) on both sides of the glass (membrane and buffer reservoir) selectively interact with the glass, setting up a potential gradient across the glass membrane. Since the hydrogen ion concentration in the internal buffer solution is invariant, this potential difference, determined relative to the Ag/AgCl reference electrode, is proportional to the pH of the media. Prior to every sampling day, the instrument was calibrated against pH reagents at 7 and one other pH value (either 4 or 10). The output from the sonde sensor was processed via the sonde software to provide pH readings.

The sonde employs an YSI Rapid Pulse system for the measurement of DO. The Rapid Pulse system utilizes a Clark-type sensor that is similar to other membrane-covered steady-state DO probes. The system measures the current associated with the reduction of oxygen, which diffuses through a Teflon membrane, and this current is proportional to the partial pressure (not the concentration) of oxygen in the solution. The membrane isolates the electrodes necessary for this reduction from the external media, encloses the thin layer of electrolyte required for the current flow, and prevents other non-gaseous, electrochemically active species from interfering with the measurement. The YSI 6600 corrects the final DO concentration for in situ temperature and salinity. DO concentrations are reported in mg/L.

For the bacterial analyses, water sample collection was carried out using a Polyvinyl Chloride (PVC) Niskin bottle, with a Teflon coated un-obstructed chamber (external springs). A General Oceanics 101005-XT Niskin bottle and GO Devil messenger were deployed during this study. The Niskin bottle was deployed to collect water samples from a depth of one meter above the seabed and one meter below the ocean surface. Seawater samples were transferred to the appropriate glass bottles and labeled with date, time, station, depth, and sampling personnel.

The water samples were stored on ice in a covered cooler until they could be delivered to the analytical laboratory. Proper chain-of-custody procedures were followed for all water samples.

For the Rhodamine WT analyses, the sampling, labeling, and storage procedure was identical to that detailed above, except the sample containers were plastic.

A helicopter was on site and overhead for approximately seven hours on Tuesday, August 20, carrying personnel from USC who were photographing the dye field using a high-resolution digital camera. The helicopter also conducted a short flight along the coast (just over one hour) on the morning of Wednesday, August 21 with USC personnel to look for any remaining traces of the dye fields. Time-series photographs of the dye fields were analyzed and, from those, concentration maps of the dye fields were produced.

Personnel were stationed along the shoreline at Stations 6N, 7.5N, 9N, 10.5N, 12N, 13.5N and 15N to collect water samples for laboratory analysis of Rhodamine WT dye. The data obtained from these samples were used in conjunction with the offshore dye data and the high-resolution aerial imagery to evaluate the behavior of the AES HBGS thermal plume with respect to how it moves, where it impinges upon the shoreline, and how much it dilutes with ambient seawater before contacting the shoreline.

Personnel were stationed continuously at Stations 6N, 9N, and 12N for the duration of the study (from 09:30 on August 20 through 15:30 on August 21, 2002). For the other stations (Station 7.5N, 10.5N, 13.5N, and 15N), personnel were stationed at those locations from 09:30 on August 20 until 19:00 that night, then again the following morning on August 21 from 08:00 through to the end of the study at 15:30 that afternoon. Personnel collected these water samples at intervals of 15 minutes from 09:30 until 20:00 on August 20, then at 30-minute intervals after that until the completion of the dye study at 15:30 on August 21, 2002.

In conjunction with the intensive in-plant sampling program that was occurring during the offshore dye study, personnel along the shoreline also collected water samples for microbial analysis at 6N, 9N, and 12N at intervals of three hours from 09:30 on August 20 until 12:30 on August 21, 2002. These samples were analyzed for total coliform, fecal coliform, and Enterococcus. Water samples were also collected at these stations (one sampling event) for microbial source tracking analyses.

Water samples for laboratory analysis of Rhodamine WT dye were collected in USC laboratory-prepared, 125 ml plastic bottles. Prior to collecting the appropriate water samples, each bottle was rinsed three times with seawater. Each sample bottle was then labeled with the date, time,

station, and the initials of the sampling personnel. The water samples were stored in a covered cooler until they could be delivered to the USC analytical laboratory.

For the bacterial analyses, water samples were collected in laboratory-prepared glass bottles containing an appropriate preservative (sodium thiosulfate). Each sample container was labeled with the date, time, station, and initials of the sampling personnel. The water samples were stored on ice in a covered cooler until they could be delivered to the analytical laboratory. Proper chain-of-custody procedures were followed for all water samples.

3.5 THERMAL PLUME MODEL

3.5.1 PURPOSE

The purpose of the thermal plume model for the AES HBGS was to provide an evaluation of the dilution characteristics of the thermal plume in the marine environment in the immediate vicinity of the outfall at the AES HBGS plant. The computer modeling results of the thermal plume were compared to the empirical results from the offshore dye study. These tools were used together to assess bacterial transport mechanisms in the local oceanographic environment of Huntington State Beach.

3.6 DILUTION ANALYSIS

3.6.1 DILUTION MECHANISMS

Reduction in contamination in an effluent plume occurs by the following:

- Mechanical mixing due to high velocity effluent discharge into low velocity ambient water;
- Aspiration of the receiving water into the discharge plume during buoyant rise;
- Lateral dispersion of the effluent as it is carried horizontally by littoral current patterns; and
- Reduction in viable bacteria due to natural die-off.

3.6.1.1 Initial Dilution (Near-Field)

The first two points noted above (mechanical mixing and dilution due to buoyant rise) are commonly referred to as initial dilution or near-field dilution. Mechanical mixing is proportional to the velocity of the discharge and therefore related to the geometry of the discharge port(s). The AES HBGS outfall is a single vertical port with a large cross-sectional area relative to the discharge volume; consequently, the relative velocity of discharge is low and mechanical mixing is minimal.

The effluent being discharged has the same salinity as the ambient ocean water, but is approximately 10 to 20°F warmer than the receiving water (as measured in the intake and discharge vaults). The temperature variation between the effluent and the receiving water decreases during the summer (May to August) when the ambient water temperature is generally higher. A warmer effluent discharged into a cooler ambient will be buoyant. When large volumes of this type of water are discharged at a high rate (220,000 gpm or higher), vertically, from shallow depths (approximately 10 feet beneath the surface), there is sufficient momentum and buoyant force to carry the plume to the surface with minimal initial dilution, regardless of ambient water column characteristics. Accordingly, there is minimal difference in initial dilution at the AES HBGS outfall throughout the year.

3.6.1.2 Dispersion (Far-Field Dilution)

Dispersion of the effluent plume occurs as the ambient currents move it laterally. This process, similar in nature to molecular diffusion, effects a further reduction in the concentration of the effluent plume. The process is conventionally quantified by a series of partial differential equations (Fisher et al., 1979).

A major component of the analysis is the selection of a diffusion coefficient, which represents the time rate of transport of the element across a unit area of the receiving medium, divided by the gradient of the concentration. Several researchers have studied this phenomenon. The most widely accepted approach (used in the USEPA models and detailed in the sections that follow) was developed by Norman Brooks. Technical documentation of these approaches can be found in the USEPA document EPA/600/R-94/086 (Baumgartner et al., 1994).

3.6.1.3 Bacterial Disappearance

When discharged into a marine receiving environment, indicator bacteria concentrations are known to decrease as a result of natural death, inactivation, coagulation, flocculation, sedimentation, or predation. The rate of loss of viability (regardless of cause) is commonly evaluated using the following formula:

$$C = Coe^{-kt}$$

Where:

C = concentration of viable organisms at any time;

Co = original concentration of organisms;

k = rate of decay coefficient; and

t = time.

The rate of decay coefficient is, by convention, eliminated by defining the elapsed time for 90% of the viable bacteria to disappear (T90).

As disappearance is dependent on several factors (one of which is exposure to UV light), T90 exhibits diurnal and seasonal variations. T90 can vary from as short as two hours when the sun is directly overhead and the plume is on the surface, to as long as 40 hours during the night. Accordingly, no single T90 is appropriate for all cases. For estimation purposes in this report, a range of T90 values have been used. The product of the T90 and the total physical dilution yields a representation of the total dilution (or reduction in concentration) of a non-conservative element such as indicator bacteria (Velz, 1984).

3.6.1.4 Dilution Analysis Model

The USEPA model UM and the Brooks far-field model were used to complete this analysis (Baumgartner et al., 1994). UM was published in June 1994 and is fully described in the technical documentation previously referenced. The model is applicable to single and multiple port diffuser configurations, and can be used in flowing or stagnant, stratified or un-stratified environments.

UM is primarily a "near-field" (initial dilution) model, using a Lagrangian formulation and the projected area entrainment hypothesis (the rate at which ambient water is entrained into the wastewater field in the presence of current).

The Brooks far-field algorithms can be implemented as a sub-program to UM. The algorithms use the plume characteristics predicted by the near-field algorithms to predict the dilution due to dispersion as the plume moves laterally (under the influence of littoral currents), after the cessation of buoyant rise.

UM also calculates the reduction in concentration of non-conservative substances (such as fecal coliform, or Enterococcus bacteria) during the time of lateral movement.

3.6.1.5 Input Variables

UM requires numerical characterization of the following:

- The discharge (flow rate, salinity, temperature, and initial contaminant concentration);
- The diffuser (diameter, orientation, depth);
- The receiving water (salinity, temperature, current speed); and
- Decay coefficient of non-conservative substances (indicator bacteria).

The input parameters used in each calculation are printed as a preface on the respective calculations included with this report (**Appendix F)**. For these modeling runs, analysis was completed using the following variables.

Discharge Characteristics

Total flow - 220,000 gpm to 352,000 gpm in 44,000 gpm increments, according to the number of pumps online (5 to 8, respectively);

Salinity - per ambient data for the given season;

Temperature - relative to ambient data; and

Initial contaminant concentration - 100 CFU/100 ml (representative of the maximum allowable single-sample Enterococcus concentration, but also to facilitate ease of calculating total dilution = $100/C_t$).

Diffuser Characteristics

Number of ports - one (1);

Port diameter - equivalent to 19.7 feet in diameter;

Vertical orientation - 90° (straight up); and

Port Depth - 13.5 feet (average port depth).

Receiving Water Characteristics

Depth / temperature / salinity profiles for May, August, and late September; and

Currents – 0.05 m/s, 0.10 m/s and 0.15 m/s.

Decay (T90) Coefficients

Four hours, eight hours and 16 hours.

The model was written for inputs in metric units. Accordingly, all inputs were converted from U.S. customary units to metric units. Additionally, the default values of the program were used which include the far-field dispersion coefficient value of 0.0003 $m^{2/3}/s$ for the Brooks equation. This value is considered to be a conservative value applicable to near-coastal waters. In the case of open coastal waters, a value of 0.000453 would be used.

3.7 MICROBIAL SOURCE TRACKING

Microbial source tracking was used on water samples collected in both the AES HBGS plant and the Blackford's Ditch. Most microbiological water quality studies have used the now-classical

assays for coliforms and Enterococcus. While these have proven to be quite useful for determining when contamination is present at a particular location, there is no source discrimination. The contamination source of highest concern comes from human waste. The role of microbial source tracking was to look specifically for a human-derived pathogenic enteroviruses, and human-specific fecal bacteria.

Sample locations for microbial source tracking were based on the first few weeks of microbial sampling. 64 samples were analyzed for human bacteria and human viruses, and were collected from the following locations:

- Intake vault;
- Discharge vault;
- Storm water sump;
- General purpose retention basin; and
- Blackford's Ditch

Samples were collected during the intensive sampling between August 20 and August 23, 2002. Microorganisms from samples were collected by filtering 300 to 1000 ml of sample water, through a 47 mm diameter, 0.22 μm pore size Durapore filter; prefilters were not used. DNA extraction can be inhibited by compounds such as humic acids; two DNA extraction methods were tested; a standard detergent-phenol extraction (hot 1% SDS lysis with phenol purification, Fuhrman et al., 1988), and a commercially available kit (Bio101 soil extraction kit). Extracted DNA concentrations were quantified by Pico Green fluorescence using a BioRad fluorometer, both methods yielded equivalent quantities of DNA. The Bio101 kit was selected because of its efficiency and ability to remove humic acid. DNA samples were split into two tubes prior to running further tests, to enable subsequent confirmation of an initial positive sample.

3.7.1 BACTEROIDES/PREVOTELLA ANALYSIS

Amplification of the human-specific *Bacteroides/Prevotella* marker followed the procedure of Bernhard and Field (2000), using PCR primers that amplify partial 16S rRNA from Human Fecal (HF) specific group. Most amplifications were from 5-15 nanograms (ng) of extracted DNA, equivalent to about 10 ml of seawater, chosen to provide an optimal compromise between sensitivity and avoidance of inhibition of the assay. All sets of assays included negative controls (no DNA added), and positive controls, in which a small amount (approximately 1 picogram [pg]) of human-fecal DNA extract was added. All samples with positive results from the first round were re-tested with DNA from the original split sample, and only those that showed positive on this second round were considered confirmed positives. The test is

extremely sensitive, with multiple-million-fold amplification, and it is possible for contamination to yield occasional false-positives, therefore the re-test eliminated the significance of potential contamination.

3.7.2 ENTEROVIRUS ANALYSIS

The "Real Time RT-PCR" method developed by Monpoeho et al (2000) was used in this study analysis of seawater samples. Viruses were collected on a filter, and the viral RNA (viral genetic material) extracted. A DNA copy of the RNA was made by reverse transcription of the mRNA template, and that DNA was amplified by PCR in an instrument that continuously monitored each sample tube for appearance of amplified products. During amplification, internal probe fluorescence developed when the PCR product accumulated above a threshold, and that fluorescence was indicative of the specific product. The PCR cycle during which the probe was detected was directly related to the amount of viral RNA in the original sample. The detection limit was approximately one plaque-forming unit of poliovirus.

Sample locations and times coincided with those for the Bacteroides/Prevotella analysis samples. Samples with volumes of 300 ml to 2000 ml were filtered through glass fiber filters (Gelman type AE), and frozen. At the USC laboratory, they were extracted with a Qiagen RNeasy mini kit, into a final volume of 50 μl. Of this, 5 ul was tested in each assay, following the protocol of Monpoeho et al. (2000), with a BioRad quantitative PCR apparatus. Each sample, tested in duplicate, had a matching pair of "spiked" samples that had a fixed amount of cultured poliovirus added. Sample sets run on a given day included negative controls and a poliovirus standard set. Results were considered positive if the probe fluorescence developed within the 50 cycles in both replicates.

3.8 MOORING OBSERVATIONS NEAR THE INTAKE AND DISCHARGE

Moored measurements of temperature and conductivity were obtained at two sites near the HBGS cooling water intake and cooling water discharge outfall during the period between August 16 and October 20, 2002. The primary purpose of these observations was to determine whether or not low-salinity, low-temperature water (sub-thermocline) came into the proximity of the cooling water intake pipe such that the low-salinity, low-temperature water could be entrained into the cooling water and then dispersed onto the beach from the cooling water discharge. The deployment locations are identified in **Plate 3-1** and **3-2.**

Each of the moorings was configured with two SB-MC, one near surface and one near the bottom of the mooring (**Figure A-4).** A summary of sensors, sensor position on mooring, and

mooring locations for the two mooring deployments between August 16 and October 20, 2002 are listed in the table below. In addition to the two SB-MC, the moorings were equipped with 8 Onset HOBO temperature sensors to resolve temperature at approximately one-meter intervals between near surface and near bottom.

The moorings were deployed for two intervals, with a brief break in observations between the intervals while the moorings were being serviced (**Table 3-3**). During each interval the sensors were programmed to sample at two-minute intervals. SB-MC were also deployed in the intake and discharge vaults of the power plant so that comparisons of the water entering and leaving the plant could be made. These SB-MC were programmed to log data at one-minute intervals.

Table 3-3 Mooring Deployment Summary

DEPLOYMENT #1 Depth (m)	**Outfall**		**8/16/2002 13:30 PDT Intake/shoreward**	**9/24/2002 08:00 PDT**
	Surface mooring	*Bottom Mooring*	*Surface mooring*	*Bottom Mooring*
0.9 below sfc	SBE 2442		SBE 2380	
1.8 below sfc	Hobo 581034		Hobo 581027	
2.7 below sfc	Hobo 581025		Hobo 581032	
3.6 below sfc	Hobo 581031		Hobo 581030	
4.5 below sfc	Hobo 581033		Hobo 581035	
4.6 above bottom		Hobo 581026		Hobo 581029
2.4 above bottom		SBE 1223		SBE 1222
1.6 above bottom		S4 05812484 SES release 48268-02		SES Release 49625-03
0.5 above bottom		Hobo 581028		Hobo 581024

DEPLOYMENT #2 Depth (m)	**Outfall**		**9/26/2002 13:30 PDT Intake/shoreward**	**10/20/2002 08:00 PDT**
	Surface mooring	*Bottom Mooring*	*Surface mooring*	*Bottom Mooring*
0.9 below sfc	SBE 2380		SBE 1629	
1.8 below sfc	Hobo 581030		Hobo 600709	
2.7 below sfc	Hobo 581035		Hobo 600707	
3.6 below sfc	Hobo 600708		Hobo 581029	
4.5 below sfc	Hobo 581024		Hobo 581032	
4.6 above bottom	-	Hobo 581027		Hobo 581026
2.4 above bottom	-	SBE 1223		SBE 1222
1.6 above bottom	-	S4 05812484 SES release 48268-02		SES Release 49625-03
0.5 above bottom	-	Hobo 581028		Hobo 600710

4 RESULTS

4.1 IN-PLANT WATER QUALITY

4.1.1 MICROBIAL AND PHYSICAL ANALYSIS BY LOCATION

Samples were collected daily during the 14 weeks of sampling between July 12 and October 5, 2002. For a two-week period between August 16 and 29, 2002, samples were collected every three hours from tidally influenced locations (intake and discharge vaults and Blackford's Ditch). During the intensive sampling, samples were collected once per day from all eight locations. The absolute values, ranges, and means for bacterial, physical, and chemical measurements collected during the two sampling periods are summarized on **Figures 4-1 to 4-32, Plates 4-1 to 4-15, and Tables 4-1 and 4-2**. Any significant variations or trends have also been identified. All reported laboratory data have been included in **Appendix G** (Sierra Analytical), **Appendix H** (Scripps Institute of Oceanography), and **Appendix I** (Severn Trent Laboratory). Bacterial concentrations have been ranked in each sample location based on an arbitrary scale (**Table 4-1**):

Table 4-1 Arbitrary Bacterial Concentration ranking

Category	Total Coliform (mean MPN/100ml)	Fecal Coliform (mean MPN/100ml)	Enterococcus (mean MPN/100ml)
Low	Less than 150	Less than 50	Less than 50
Moderate	150 – 2,500	50 – 250	50 – 500
Moderately High	2,500 – 10,000	250 – 500	500 – 2,500
High	10,000 or greater	500 or greater	2,500 or greater

4.1.1.1 Intake Vault

Bacterial concentrations in the IV were generally low (**Figures 4-1a and 4-1b**). Total coliform concentrations ranged from below detection limits to 310 cfu/100ml, with a mean of 11 cfu/100ml. Fecal coliform concentrations ranged from below detection limits to 110 cfu/100ml, with a mean of two cfu/100ml. Enterococcus concentrations ranged from below detection limits to 590 cfu/100ml, with a mean of 17 cfu/100ml. The ratio of total coliform to fecal coliform to Enterococcus was generally 11:2:17. The bacterial values observed in the IV sample location would have represented a total of four exceedances of AB 411 had the samples been collected

from the surf zone **(Plates 4-1 to 4-15)**. Temperature varied from 14 to 23.5°C, with a mean of 18.2°C. Field salinity varied from 17.3 to 35.1 ppt, with a mean of 28.14 ppt. pH varied from 7.2 to 8.6, with a mean of 8.1. Conductivity varied from 28.2 to 52.7 µS/cm, with a mean of 43.6 µS/cm. Turbidity fluctuated from 0 to 214 nephelometric turbidity units (NTU), with a mean of 15.3 NTU, and DO varied from 4.8 to 12.6 mg/L, with a mean of 9.2 mg/L **(Tables 4-1 and 4-2).**

4.1.1.2 Discharge Vault-0

Bacterial concentrations at DV_0 (zero feet below water surface) were low (**Figure 4-2a and 4-2b**). Total coliform concentrations ranged from below detection limits to 1,500 cfu/100ml, with a mean of 136 cfu/100ml. Fecal coliform concentrations ranged from below detection limits to 420 cfu/100ml, with a mean of 12 cfu/100ml. Enterococcus concentrations ranged from below detection limits to 470 cfu/100ml, with a mean of 16 cfu/100ml. The ratio of total coliform to fecal coliform to Enterococcus was generally 10:1:1. The bacterial values observed in the DV_0 sample location would have represented a total of six exceedances of AB 411 had the samples been collected from the surf zone **(Plates 4-1 to 4-15)**. Temperature varied from 16.0 to 32.8°C, with a mean of 23.8°C. Field salinity varied from 1.1 to 40.0 ppt, with a mean of 27.5 ppt. pH varied from 6.4 to 8.5, and a mean of 8.0. Conductivity varied from 3.5 to 69.0 µS/cm, with a mean of 40.4 µS/cm. Turbidity varied from 0 to 387 NTU, with a mean of 22.9 **(Tables 4-1 and 4-2)**.

4.1.1.3 Discharge Vault-10

Bacterial concentrations at DV_{10} (approximately ten feet below water surface, however because of tidal influence in the discharge vault samples for DV_{10} were collected at the entrance to the discharge pipe) were low (**Figure 4-3a and 4-3b**). Total coliform concentrations ranged from below detection limits to 1,700 cfu/100ml, with a mean of 93 cfu/100ml. Fecal coliform concentrations ranged from below detection limits to 180 cfu/100ml, with a mean of 46 cfu/100ml. Enterococcus concentrations ranged from below detection limits to 610 cfu/100ml, with a mean of 12.2 cfu/100ml. The ratio of total coliform to fecal coliform to Enterococcus was approximately 20:1:2. The bacterial values observed at DV_{10}would have represented a total of three exceedances of AB 411 had the samples been collected from the surf zone **(Plates 4-1 to 4-15)**. Temperature varied from 17.5 to 32.1°C, with a mean of 24.1°C. Field salinity varied from 4.3 to 40ppt, with a mean of 27.5ppt. Conductivity varied from 3.5 to 70.5 µS/cm, with a mean of 42.0 µS/cm. pH varied from 6.7 to 8.5, with a mean of 8.0. Turbidity varied from 0 to 240 NTU, with a mean of 17.5 NTU **(Tables 4-1 and 4-2).**

4.1.1.4 Discharge Vault-30

A limited set of samples was collected at DV_{30} (a depth of 30 feet below ground surface) in the discharge vault (**Figure 4-4**) from September 18, to October 15, 2002. Fecal coliform and Enterococcus concentrations were generally below detection limits (<1 cfu/100 ml), total coliform concentrations were predominantly less than 10 cfu/100 ml, with only three values (40,20 and 20) greater than 10 cfu/100ml. The bacterial concentrations observed at DV_{30} are similar to concentrations observed at DV_{10} during the same time period, with the exception of occasionally higher total coliform concentrations.

4.1.1.5 Opposite Flow

Samples were collected from OF on six separate occasions during the daily sampling, all samples were collected between 09:08 and 09:55. Sample dates were August 6 through August 10, and September 26, 2002. Total coliform concentrations ranged from 20 to 200 cfu/100ml; fecal coliform concentrations ranged from <1 to 12 cfu/100ml and Enterococcus concentrations <1 to 4 cfu/100ml. Temperature at OF during sampling ranged from 22°C (72.0 °F) to 29.0°C (84.2 °F).

4.1.1.6 Blackford's Ditch

Bacterial concentrations in BD were high (**Figure 4-5a and 4-5b**). Total coliform concentrations ranged from 200 to 370,000 cfu/100ml, with a mean of 18,311 cfu/100ml. Fecal coliform concentrations ranged from below detection limits to 31,000 cfu/100ml, with a mean of 825 cfu/100ml. Enterococcus concentrations ranged from 10 to 62,000 cfu/100ml, with a mean of 4,824 cfu/100ml. The ratio of total coliform to fecal coliform to Enterococcus was generally 20:1:5. The bacterial values observed in the BD sample location would have represented a total of 312 exceedances of AB 411 had the samples been collected from the surf zone **(Plates 4-1 to 4-15)**. Temperature varied from 13.4 to 28.1°C, with a mean of 21.3°C. Field salinity varied from 1.0 to 23.20 ppt with a mean of 10.00 ppt. pH varied from 7.1 to 9.4, and a mean of 8.2. Conductivity varied from 2.2 to 36.6 µS/cm, with a mean of 16.9 µS/cm. Turbidity varied from 1 to 700 NTU, with a mean of 35.3 NTU, and DO varied from 0.1 to 10.9 mg/L, with a mean of 2.9 **(Tables 4-1 and 4-2)**.

4.1.1.7 Boiler Fireside Wash

Bacterial concentrations in the boiler fireside wash were high (**Figure 4-6**). Total coliform concentrations ranged from 110 to 68,000 cfu/100ml, with a mean of 7,964 cfu/100ml. Fecal

coliform concentrations ranged from two to 4,300 cfu/100ml, with a mean of 545 cfu/100ml. Enterococcus concentrations ranged from 120 to 160,000 cfu/100ml, with a mean of 10,177 cfu/100ml. The ratio of total coliform to fecal coliform to Enterococcus was generally 16:1:20. The bacterial values observed in the BFW sample location would have represented a total of 196 exceedances of AB 411 had the samples been collected from the surf zone **(Plates 4-1 to 4-15)**. Temperature varied from 16.0 to 24.9°C, with a mean of 18.6°C. Field salinity varied from 0.7 to 13.8 ppt, with a mean of 3.8 ppt. pH varied from 5.0 to 8.1, with a mean of 6.9. Conductivity varied from 1.5 to 22.7 µS/cm, with a mean of 7.0 µS/cm. Turbidity varied from 0 to 125 NTU, with a mean of 35.4 NTU, and DO varied from 0.0 to 10.4 mg/L, with a mean of 4.1 mg/L **(Tables 4-1 and 4-2)**.

4.1.1.8 Boiler Sump Wash

Bacterial concentrations in the BSW were moderate (**Figure 4-7**). Total coliform concentrations ranged from ten to 21,000 cfu/100ml, with a mean of 1,520 cfu/100ml. Fecal coliform concentrations ranged from below detection limits to 3,800 cfu/100ml, with a mean of 228 cfu/100ml. Enterococcus concentrations ranged from 20 to 11,000 cfu/100ml, with a mean of 562 cfu/100ml. The ratio of total coliform to fecal coliform to Enterococcus was generally 15:2:5. The bacterial values observed in the BSW sample location would have represented a total of 76 exceedances of AB 411 had the samples been collected from the surf zone **(Plates 4-1 to 4-15)**. Temperature varied from 25.0 to 29.4°C, with a mean of 26.5°C. Field salinity varied from 0 to 0.8 ppt, with a mean of 0.5 ppt. pH varied from 6.4 to 7.8, with a mean of 7.2. Conductivity varied from 0.0 to 1.8 µS/cm, with a mean of 1.1 µS/cm. Turbidity varied from 1 to 999 NTU, with a mean of 300.8 NTU, and DO varied from 6.5 to 6.8 mg/L, with a mean of 6.6 mg/L **(Tables 4-1 and 4-2).**

4.1.1.9 Storm Water Sump

Bacterial concentrations in the SWS were moderate (**Figure 4-8**). Total coliform concentrations ranged from below detection limits to 11,000 cfu/100ml, with a mean of 966 cfu/100ml. Fecal coliform concentrations ranged from below detection limits to 1,400 cfu/100ml, with a mean of 92 cfu/100ml. Enterococcus concentrations ranged from below detection limits to 2,200 cfu/100ml, with a mean of 138 cfu/100ml. The ratio of total coliform to fecal coliform to Enterococcus was generally 10:1:1. The bacterial values observed in the SWS sample location would have represented a total of 25 exceedances of AB 411 had the samples been collected from the surf zone **(Plates 4-1 to 4-15)**. Temperature varied from 19.6 to 30.4°C, with a mean of 24.1°C. Field salinity varied from 0 to 8.7 ppt, with a mean of 0.8 ppt. pH varied from 6.9 to 9.7,

with a mean of 7.8. Conductivity varied from 0.0 to 14.9 µS/cm, with a mean of 1.7 µS/cm. Turbidity varied from 0 to 999 NTU, with a mean of 25.3 NTU, and DO varied from 0.0 to 9.0 mg/L, with a mean of 5.1 mg/L **(Tables 4-1 and 4-2)**.

4.1.1.10 General Purpose Retention Basin

Bacterial concentrations in the GP were moderate (**Figure 4-9**). Total coliform concentrations ranged from below detection limits to 32,000 cfu/100ml, with a mean of 2,050 cfu/100ml. Fecal coliform concentrations ranged from below detection limits to 9,400 cfu/100ml, with a mean of 101 cfu/100ml. Enterococcus concentrations ranged from below detection limits to 700 cfu/100ml, with a mean of 64 cfu/100ml. The ratio of total coliform to fecal coliform to Enterococcus was generally 20:1:1. The bacterial values observed in the GP sample location would have represented a total of 35 exceedances of AB 411 had the samples been collected from the surf zone **(Plates 4-1 to 4-15)**. Temperature varied from 20.7 to 29.8°C, with a mean of 24.8°C. Field salinity varied from 0.6 to 15.7 ppt, with a mean of 4.3 ppt. pH varied from 7.2 to 9.2, with a mean of 8.3. Conductivity varied from 1.4 to 25.5 µS/cm, with a mean of 7.8 µS/cm. Turbidity varied from 1 to 126 NTU, with a mean of 14.3 NTU, and DO varied from 1.2 to 10.3 mg/L, with a mean of 6.2 mg/L **(Tables 4-1 and 4-2)**.

4.1.2 SALINITY ANALYSIS

The salinity profile in the discharge vault was analyzed by taking samples at one-foot intervals from the surface of the water to a depth of up to 30.5 feet. Samples were collected via a pump inserted into the stainless steel well screen placed in the discharge vault. Field measurements of salinity were collected using a Horiba U-10 meter. Approximately 1,000 samples were collected over a 45-day period. Salinity in the discharge vault was recorded as varying from 17.4 to 34.6 ppt (**Table 4-3**). Low salinity values were attributed to drift on the field salinity equipment and were checked against salinity values obtained in the IV and discharge vault using the more reliable and accurate SB-MC, and from field samples analyzed by Sierra Analytical and Scripps Institute of Oceanography. The change in salinity over depth was calculated and presented as ΔSalinity (**Table 4-3**), ensuring that despite a drift in the absolute value of salinity, a comparative assessment could still be derived. After determining the questionable low values, salinity in the discharge vault was plotted against depth (**Figure 4-10**) with the omission of the values that were confirmed as unreliable. Average salinity in the discharge vault increased with depth from 33.5 ppt at the surface to 34.0 ppt at 20 to 30 foot depth, with an apparent mixing zone in the upper eight feet of the water column. Salinity at the surface ranged from 31.6 to 34.0 ppt, mid-range (20 feet) from 33.4 to 34.7ppt, and at 30 feet from 33.4 to 34.6 ppt.

Salinity analysis was conducted by Sierra Analytical and Scripps Institute of Oceanography at zero-feet, ten-feet, and 30-feet depths. Salinity at zero feet averaged 31.421 ppt, with a range of 3.331 to 34.40 ppt; salinity at ten feet averaged 32.949 ppt, with a range of 30.60 to 34.20 ppt; salinity at 30 feet averaged 33.238 ppt with a range of 30.8 to 36.5 ppt **(Table 4-4).**

4.1.3 AMMONIA ANALYSIS

Ammonia samples were collected from four locations (discharge vault at zero-feet, ten-feet, and 30-feet depths, and the IV). Ammonia concentrations in the discharge vault at zero-feet averaged 1.91 with a range of 1.20 to 3.65 mg/L; ammonia concentrations in the discharge vault at ten-feet averaged 1.71 with a range of 1.00 to 3.55 mg/L; ammonia concentrations in the discharge vault at 30-feet averaged 1.85 with a range of 1.05 to 2.80 mg/L. Ammonia concentrations in the IV averaged 2.20 with a range of 1.00 to 3.90 mg/L **(Table 4-4).**

4.1.4 STATISTICAL ANALYSIS BY ANALYTE

Analytical data collected during this study was analyzed using Statistica (StatSoft, Inc. 1995). Descriptive summary statistics (*e.g*. mean, standard deviation, variance, range, confidence intervals) for each parameter at each location are presented in **Appendix J** (**Table J-1 to J-11**). The parameters collected during this study had a high degree of variability and, after preparing probability plots (expected versus observed values), were not normally distributed (except for temperature). Therefore, a log transformation was applied to the raw data **(Table J-12)**. Water quality parameters measured in the study were graphed showing the mean, standard deviation, standard error, outliers, and extremes. The ranges of outliers and extremes are illustrated in the "box and whisker" plots (**Figure 4-11)**. The upper/lower box values represent the mean +/- the standard error, while the upper/lower whisker values present the mean +/- the standard deviation. Data points were deemed as outliers if the values were between one and a half to three times the range of the standard error. Data points were deemed as extreme if the values were greater than three times the range of the standard error.

Data were further evaluated using an analysis of variance (ANOVA) test for the significant differences between means. The purpose of ANOVA is to test the differences in means (for groups or variables) for statistical significance, accomplished by analyzing the variance (by partitioning) the total variance into the component that is due to true random error (*i.e*. within-group) and the components that are due to differences between means. These latter variance components are then tested for statistical significance, and, if significant, the hypothesis of no differences between means is rejected, and the alternative hypothesis that the means (in the population) are different from each other is accepted.

The assumptions for using ANOVA are that the data points are independent and normally distributed. However, after testing the analytical data (with the exception of temperature), it was determined that the data were not normally distributed. Therefore, the data were log transformed, resulting in a normal distribution.

In order to test for statistical differences between means, the Tukey's test for unequal sample size was performed **(Appendix J Table J-13).** While there are a number of multiple comparison procedures that compute the differences between multiple population means (Tukey's, Fisher's, Student-Newman-Keul's, Duncan's, and Scheffe's methods), the Tukey's method best accounts for unequal sample size. Tukey's procedure also allows for the construction of simultaneous confidence intervals for all pairs of treatment differences. Discussions of post-hoc procedures provided by Winer (1962), Hays (1988), or Milliken and Johnson (1984). For specific information on the Tukey's test for evaluation of data with unequal samples sizes, see Spjotvoll and Stoline (1973).

The data from each analytical parameter evaluated in the study were also graphed (**Figures 4-12 to 4-22**). The descriptive summary statistics for each parameter at each location are presented in **Table J-12**. The parameters collected during this study had a high degree of variability and, after preparing probability plots (expected versus observed values), were not normally distributed (except temperature). Normal probability plots for the transformed data and temperature are presented on **Figures 4-23 to 4-33**.

4.1.4.1 Total Coliform Bacteria

The freshwater sampling locations (BD, BFW, BSW, GP and SWS) had higher total coliform concentrations than the saltwater locations (OF, IV, DV_0, DV_{10} and DV_{30}) (**Table 4-1 and 4-2**). BD had the highest concentration of total coliform measured and BFW had the second highest concentration of total coliform. BSW, GP, and SWS had statistically equal bacteria concentrations, and had the third highest total coliform concentrations (**Table J-1 to J-11**). When OF is considered, all of the saltwater locations (OF, IV, DV_0, DV_{10} and DV_{30}) were statistically equal (**Table J-13**). When OF is not considered, the concentration of total coliform is not considered statistically equal between the IV and the discharge vault at 10 feet deep (DV_{10} > IV). All locations generally had many outliers and extremes resulting from values much higher and lower than the mean (**Figure 4-12**).

4.1.4.2 Fecal Coliform Bacteria

The freshwater locations (BD, BFW, BSW, GP, and SWS) had higher fecal coliform concentrations than the saltwater locations OF, IV, DV_0, DV_{10} and DV_{30} (**Table J-1 to J-11**). BD was found to have the greatest concentration of fecal coliform and BFW had the second highest concentration of fecal coliform. BSW, GP, and SWS had statistically equal fecal coliform concentrations, and had the third highest fecal coliform concentrations (**Table J-1 to J-11**). When OF is considered, all of the saltwater locations (OF, IV, DV_0, DV_{10} and DV_{30}) were statistically equal concentrations of fecal coliform (**Table J-13**). When OF is not considered, the concentration of fecal coliform is not considered statistically equal between the IV and the discharge vault at ten-feet deep (DV_{10} > IV). All locations generally had many outliers and extremes resulting from values much higher and lower than the mean (**Figure 4-13**).

4.1.4.3 Enterococcus

The freshwater locations (BD, BFW, BSW, GP, and SWS) had higher Enterococcus concentrations than the saltwater locations OF, IV, DV_0, DV_{10} and DV_{30} (**Table J-1 to J-11**). BFW had the greatest concentration of Enterococcus, BD the second highest concentration and BSW had the third highest concentration of Enterococcus. GP and SWS were statistically equal Enterococcus concentrations **(Table J-13)**. When OF is considered, all of the saltwater locations (OF, IV, DV_0, DV_{10} and DV_{30}) were statistically equal concentrations of Enterococcus (**Table J-13**). When OF is not considered, the concentrations of Enterococcus were also considered statistically equal between the IV and the discharge vault at ten-feet deep (DV_{10} = IV). All locations generally had many outliers and extremes resulting from values much higher and lower than the mean (**Figure 4-14**).

4.1.4.4 pH

The pH values of all the locations varied significantly over the course of the study. BD had the highest measured pH of 9.5, while BFW had the lowest measured pH of 5.0 (**Table J-1 to J-11**). The mean pH values for BD, GP, SWS, OF, IV, DV_0, DV_{10} and DV_{30} were tending towards alkalinity (pH greater than 7.5). The mean pH values for BFW and BSW were much closer to neutral (pH of 7.0). The pH measurements varied widely for each location with a maximum range of 2.4 pH units for BD and a minimum range of 0.2 pH units for OF. The mean pH values of the saltwater samples were statistically equal (**Table J-13).** All locations generally had many outliers and extremes resulting from values much higher and lower than the mean (**Figure 4-15**).

4.1.4.5 Dissolved Oxygen

Dissolved oxygen was measured only in the freshwater samples and the Intake Vault. Dissolved oxygen analysis was not conducted in the discharge vault samples. Samples from BD had the greatest range of variation (largest maximum [10.9 mg/L] and smallest measurable minimum [0.1 mg/L]) (**Table J-1 to J-11**). The mean DO concentrations of samples from the IV were higher than the DO concentration of samples collected from the freshwater samples (**Table J-1 to J-11**). The DO of the IV was statistically equal to BFW, SWS, and OF (**Table J-13**). All locations generally had many outliers and extremes resulting from values much higher and lower than the mean (**Figure 4-16**).

4.1.4.6 Salinity

The field-measured salinity using the Horiba U-10 produced some highly variable data. The questionable data has been removed from the data set, and the remaining data set has to be considered in this context. For assessment of salinity stratification in the discharge vault, data from the SeaBird MicroCats and Scripps salinity assessment has been used. The field-measured salinity data does demonstrate that:

- The freshwater locations BD, BFW, BSW, GP, and SWS had lower mean field-measured salinity concentrations than the saltwater locations OF, IV, DV_0, DV_{10} and DV_{30}; and
- The maximum field salinity measurements were sampled from IV and DV. The minimum field salinity measurements were taken at locations BD, BFW, BSW, SWS and GP.

The laboratory-measured salinity was similar for all saltwater locations and used to assess DV salinity profiles. (**Figure 4-10 and 4-17)**.

4.1.4.7 Temperature

Temperatures at all sampling locations varied widely (a range of 4.4°C [25°C to 29.4°C] at location BSW to 16.8°C [19.6°C to 36.4°C] for SWS). Temperatures in the fresh water locations remained less than 37°C, while temperatures in the saltwater locations were less than 33°C. The GP and boiler sump wash locations had the highest mean temperatures for the fresh water locations. The mean temperature for the saltwater samples was found to be statistically equal (**Table J-13**). The IV and SWS locations had the statistically lowest temperatures (**Figure 4-18**).

4.1.4.8 Turbidity

The fresh water locations BSW and SWS had the highest measured turbidity values. The highest mean turbidity value was measured at location BSW. Due to the wide range of variability in the turbidity measurements, there is no significant statistical difference between the fresh water and saltwater locations (**Table J-1 to J-11**). All locations generally had many outliers and extremes resulting from values much higher and lower than the mean (**Figure 4-19**).

4.1.4.9 Ammonia

BFW had the highest ammonia concentration measured (24.2 mg/L) and the greatest range of concentrations measured (1.6 mg/L to 24.2 mg/L). The mean value of the ammonia concentration measured at BFW was statistically different from the samples collected at all other locations (fresh water and saltwater). The mean value of ammonia concentration measured at IV was statistically higher than the mean values measured at DV_0 and DV10 (**Table J-13**). DV_0, DV_{10}, and DV_{30} had many outliers and extremes resulting from values much higher and lower than the mean (**Figure 4-20**). The ammonia concentrations are discussed further in **Section 5.5**

4.1.4.10 Conductivity

Location DV_{10} had the maximum conductivity measured (70.5 us/cm). The mean conductivity measured in the saltwater locations (OF, IV, DV_0, DV_{10} and DV_{30}) was significantly higher than the conductivity measured in the fresh water locations (BD, BFW, GP, and SWS) (**Table J-1 to J-11**). All locations generally had many outliers and extremes resulting from values much higher and lower than the mean (**Figure 4-21**).

4.1.5 SPECIFIC CORRELATION COEFFICIENTS

Pearson correlation coefficients and p-values were determined for all possible variables and pairs. Correlation is a measure of the relation between two or more variables. The measurement scales used should be at least interval scales, but other correlation coefficients are available to handle other types of data. Correlation coefficients can range from -1.00 to +1.00. The value of -1.00 represents a perfect negative correlation, while a value of +1.00 represents a perfect positive correlation. A value of 0.00 represents a lack of correlation. The most widely used type of correlation coefficient is Pearson r, which was used in these analyses (StatSoft, Inc. 1995).

The following relationships were evaluated using Pearson Correlation coefficients.

4.1.5.1 IV Temperature and Bacteria Concentration

The relationship between total coliform, fecal coliform, and Enterococcus concentrations was not significantly correlated to IV temperature (**Table J-1 to J-11**).

4.1.5.2 IV, Discharge Vault and BD Bacterial Concentrations and Tide Height

The relationship between tide height at IV and fecal coliform and Enterococcus concentrations was found to be statistically significant (p=0.056 and p= 0.046, respectively) (**Table J-1 to J-11**).

The relationship between tide height at DV_0 and total coliform and fecal coliform concentrations was found to be inversely related and statistically significant (p=0.003 and p=0.003, respectively) (**Table J-1 to J-11**).

The relationship between tide height at DV_{10} and total coliform and fecal coliform concentrations was found to be inversely related and statistically significant (p=0.00 and p=0.007, respectively) (**Table J-1 to J-11**).

The relationship between tide height at DV_{30} and total coliform, fecal coliform and Enterococcus concentrations was not found to be statistically significant (**Table J-1 to J-11**).

The relationship between tide height at BD and total coliform, fecal coliform, and Enterococcus concentrations was found to be inversely related and statistically significant (p=0.023, p=0.067 and p=0.083, respectively) (**Table J-1 to J-11**).

4.1.5.3 IV, Discharge Vault and BD Bacterial Concentrations and Tide Direction

The relationship between tide direction at IV and total coliform, fecal coliform, and Enterococcus concentration was not found to be statistically significant (**Table J-1 to J-11**).

The relationship between tide direction at DV_0 and fecal coliform concentration was found to be positively correlated (high tide, high bacterial concentrations) and statistically significant (r^2=0.129, p=.073) (**Table J-8**).

The relationship between tide direction at DV_{10} and total coliform, fecal coliform, and Enterococcus concentrations was not found to be statistically significant (**Table J-9**).

The relationship between tide direction at DV_{30} and Enterococcus concentration was found to be positively correlated (high tide, high bacterial concentrations) and statistically significant (r^2= 0.359, p=0.015) (**Table J-10**).

The relationship between tide direction at BD and total coliform, fecal coliform, and Enterococcus concentrations was not found to be statistically significant (**Table J-1 to J-11**).

4.1.5.4 DV_0 and DV_{10} Bacterial Concentrations and Laboratory Salinity

The relationship between laboratory-measured salinity and fecal coliform, and Enterococcus concentrations at DV_0 was found to be inversely related and statistically significant (r^2 = 0.426 p=.034; and r^2= 0.590, p=.002, respectively) (**Table J-1 to J-11**).

The relationship between laboratory-measured salinity and total coliform and Enterococcus concentrations at DV_{10} was found to be inversely related and statistically significant (r^2= 0.224, p=0.002; and r^2= 0.121 p=.090, respectively) (**Table J-1 to J-11**).

4.1.5.5 IV and DV_{10} Bacterial Concentrations and Number of Pumps Operating

The relationship between the number of pumps working and Enterococcus concentration at DV_{10} was found to be statistically significant (r^2= 0.162, p=.025) (**Table J-1 to J-11**). All other relationships were not statistically significant.

4.1.5.6 IV and DV_{10} Bacterial Concentrations and Temperature at DV_{10}

The relationship between discharge temperature and fecal coliform at IV was negatively correlated and statistically significant (r^2=0.15, p=0.034) (**Table J-1 to J-11**). The relationship between discharge temperature and Enterococcus at DV_{10} was statistically significant (r^2=0.190, p=0.08) (**Table J-1 to J-11**).

4.2 BLACKFORD'S DITCH INVESTIGATION

Physical and chemical parameters were collected on a transect down Blackford's Ditch on three occasions **(Table 4-5 and Figure 4-22)**. Two transects were conducted prior to the first rain event (November) on August 13 and September 9, 2002; the third transect (after the rain event) was conducted on November 26, 2002. The results fall into two distinct categories; pre- and post-rain event.

Pre-rain event salinity at the head of the creek by the pump house ranged from 15.4 (September 9, 2002) to 22.7 ppt (August 13, 2002). On August 13, 2002, the transect was conducted over 260 feet to obtain a general perspective on conditions along the length of the creek. Salinity decreased steadily with distance from the pump house down to a low of 1.0 ppt at a distance of 260 feet; conductivity matched salinity along the length of the transect. By comparison, temperature (18.9 to 25.0°C), pH (7.61 to 9.03), turbidity (12 to 101 NTU), and DO remained relatively constant across the ditch, except at a distance of 60 feet, where DO dropped significantly from 4.43 to 0.80 mg/L. A comparatively low pH was also observed at this distance. Consequently, the subsequent transect (September 9) focused around the first 100 feet. Initial salinity was lower than the previous analysis, and remained fairly constant for the first 40 feet. At 60 feet the salinity halved from 15.8 to 7.3 ppt and remained constant over the next 40 feet. Conductivity matched salinity at all data points. Dissolved oxygen, pH, and temperature were lower on the second sampling event, with little obvious trending across the ditch.

The post-rain event transect (November 26, 2002) was markedly different from the previous two transects. During the rain event over the weekend of November 9 and10, 2002, the pump in the pump house was activated. Floating mats of macroscopic algae had significantly reduced in density by November 26, 2002 (>90% coverage pre-November 9, to <10% coverage post-November 9, 2002). A mid-range transect was conducted from the pump house to a distance of 180 feet. Initial salinity at the pump house was 2.6 ppt and did not vary significantly up to 180 feet from the pump house (3.3 ppt). Conductivity matched salinity again. pH was equivalent to previous transects ,and temperature was between 1 and 5°C lower. Turbidity was highest at 180 feet.

In addition to the transect analysis, samples were collected from Blackford's Ditch to analyze for the presence of a corrosion inhibitor used within the AES HBGS. On October 14, 2002, samples were analyzed for the presence of total borate and nitrite. On November 26, 2002, samples were analyzed for the presence of total borate, nitrite, and tolytriazole **(Table 4-6).** Total borate, nitrite and tolytriazole are all components of the corrosion inhibitor used within the AES HBGS.

On October 14, 2002, total boron and nitrite samples were collected from Blackford's Ditch at distances of 0, 30, 60, 90, 120, and 150 feet from the pump house. Boron concentrations were reported within a range of 60.2 to 72.2 mg/L, increasing slightly with distance from the pump house. Nitrite concentrations ranged from 25.2 to 15.7 mg/L, decreasing slightly with distance from the pump house. Because of the small sample number, it is not possible to determine whether there is a statistically significant difference between sample locations. An analytical

method for tolytriazole had not been identified by October 14, 2002, and consequently no analysis was conducted for this compound on this date.

A second set of analyses was conducted on Blackford's Ditch after the rain event of November 8 to 12, 2002, on November 26, 2002. Samples were collected from Blackford's Ditch at distances of 0, 20, 40, 60, 120, and 180 feet from the pump house. Boron concentrations were significantly lower than the previous sampling event, ranging from 3.8 mg/L at a distance of 0 feet, to 11.9 mg/L at a distance of 180 feet. From 0 to 120 feet, the concentration of boron remained around 4 mg/L. Nitrite concentrations were significantly lower than the previous sampling event, although matrix interference of the samples prevented the laboratory from quantifying each sample; the results were reported as not detected above a reporting limit of 10.0 mg/L. Tolytriazole concentrations ranged from below the detection limit to 'trace' amounts of 2.0 mg/L.

In summary, prior to the November rain event, salinity steadily decreased along the length of the ditch from the pump house to the dry inlet to the ditch. Salinity concentrations were approximately two-thirds the concentrations observed in ocean water. Temperature, pH, turbidity and DO remained constant across the length of the ditch. After the storm event of November, there was no change in salinity concentration along the length of the ditch.

4.3 STORM WATER SAMPLING

A storm event occurred after the cessation of the long-term in-plant sampling. The storm persisted from November 8 to November 12, 2002, depositing over two inches of rainfall in the Los Angeles Basin. The storm was the first of the season and the only significant precipitation event since the previous January. Consequently, the winter storm provided an opportunity to quantify bacterial concentrations in the sample locations during a storm event. Microbial, physical and chemical samples were collected from two depths in the discharge vault, the IV, the general purpose retention basin, Blackford's Ditch, the storm drain adjacent between the PCH and the mobile home park, and the car park drain in the wildlife sanctuary. During the field sampling, the pump located at Blackford's Ditch was activated (by a float switch) and started pumping water from the ditch to the discharge vault and ultimately the ocean.

During a site walk on November 9, 2002, field staff observed a significant amount of standing water on either side of Newland Street. A sheen (potentially hydrocarbon material) was observed flowing from the automotive wrecking facility on Edison Avenue, across the avenue into the gutter, and down the gutter to Newland Street, where the sheen flowed into the top of Blackford's Ditch. Dogs were being walked along the street and a large amount of canine fecal

material was observed alongside Newland Street and Edison Avenue. Storm water was observed running off the tank farms to the north of the AES HBGS into Edison Avenue and down into Newland street and the Blackford's Ditch.

Microbial, physical, and chemical results are presented in **Table 4-7**. Surface water quality had deteriorated significantly after the first rain event of the season in comparison to water quality during the three month sample-preiod, with total coliform concentrations approximately 20-fold and 6-fold higher than mean concentrations in Blackford's Ditch and the general purpose retention basin, respectively; fecal coliform concentrations 25- fold and 2-fold higher than mean concentrations in Blackford's Ditch and the general purpose retention basin, respectively; and Enterococcus concentrations equivalent and seven-fold higher than mean concentrations in Blackford's Ditch and the general purpose retention basin, respectively. Samples collected from both the wildlife sanctuary car park and the storm drain adjacent to the PCH had a concentration of microorganisms similar to Blackford's Ditch.

A heat-treatment process coincided with the storm event sampling on November 10, 2002. Temperatures up to 47°C were documented in the IV and OF sample locations. Because the cooling water system was undergoing a heat treatment process, the system flow had been reversed and the discharge vault did not represent the discharge point for cooling water; samples collected from OF represented discharge samples. Temperatures in the discharge vault at 0 feet and 10 feet were also elevated (32°C and 37°C for zero-feet and ten-feet depths). Both the incoming water and outgoing water exceeded AB 411 standards. Concentrations in the discharge vault were higher at zero feet than at ten feet for all three microbial analytes.

4.4 DYE TEST SANITARY SEWER

Figure 4-35 shows a time-series plot of Rhodamine WT dye concentration versus time for the 24-hour study period as measured using the Seapoint fluorometer and as logged by the acquisition software on the laptop computer. The data presented demonstrates that during the 24-hour study period at no time was Rhodamine WT dye observed to pass through the discharge vault. A review of the 24-hour-long video record also confirms that Rhodamine WT dye was not observed in the discharge vault after injection via the toilets into the sanitary sewer system. The figure also shows a typical frame from the video record. The brightness in the image is specular reflection of light from the video light and the sky above the water surface. The sampling crews collecting water samples from the seven onsite and offsite locations did not observe Rhodamine WT dye at any of those sampling locations at any time during the 24-hour study period.

4.5 NEARSHORE OCEAN ENVIRONMENT

4.5.1 PHYSICAL OCEANOGRAPHIC PARAMETERS

4.5.1.1 Differences in CTD Data Sets Between Instruments

The CTD data obtained by Komex during the offshore dye study on August 20 and 21, 2002, varied from the other CTD data sets obtained from USC offshore moorings MO1 and MO3, as well as the data obtained from CTD instruments installed in the IV and the DV of the AES HBGS. The two principal differences were as follows:

- In comparing temperature data from the USC offshore mooring MO1 and the nearest Komex offshore CTD cast (Cast #11), Komex temperatures were approximately 0.9 °C lower than those recorded at the mooring; and
- The salinity data (as calculated from temperature, conductivity and pressure data) in the vicinity of the HBGS outfall structure, as measured during CTD Cast #1, showed very low surface and near-surface salinity values (in the order of approximately 3 to 4 parts per thousand [ppt] lower than the surrounding ambient salinities and the salinities measured in the IV and DV).

Hydrographic station positions were designed to provide vertical profiles at the location of the intake and outfall and to provide water column profiles along the coast at locations to coincide with dye tracer sampling. Positions were chosen to cover north and south of the outfall/intake. The vessel was tracking the dye plume.

Minor differences in temperature and salinity data may have occurred because of equipment sensor differences related to the makes and models of the various CTD instruments. For example, the Komex offshore CTD data was gathered using an YSI 6600, whereas the IV and DV CTD data was collected using SB-MC. The sections below discuss rectification of the temperature data and the anomalously low salinities measured during offshore CTD Cast #1.

4.5.1.2 Rectification Of Offshore CTD Cast Temperature Data

As noted above, the Komex CTD cast temperature data appeared to be approximately 0.9 °C lower than temperature data recorded at USC offshore moorings. Rectification of the temperature data set for the Komex CTD data was performed by applying a linear temperature offset to the Komex temperature data. The temperature difference noted above was added to the Komex data, and this adjusted temperature data was used to recalculate salinities using the Practical Salinity Scale (PSS). The adjusted temperature data combined with the original

conductivity data and the original pressure data was input to the PSS algorithm to generate adjusted salinities. An example of the original and adjusted temperatures and salinities compared to the USC data is detailed below.

Description	T (°C)	S (ppt)
Komex (Original Data)	20.930	34.530
USC Mooring Data	21.814	33.633
Delta	**-0.884**	**0.897**
Komex (Adjusted Data)	21.814	33.611
USC Mooring Data	21.814	33.633
Delta	**0.000**	**-0.022**

4.5.1.3 Surface Salinity Data from Cast #1

The salinity data measured during CTD Cast #1, in the vicinity of the HBGS outfall structure, showed very low surface and near-surface salinity values (approximately 3 to 4 parts per thousand [ppt] lower than the surrounding ambient salinities and the salinities measured in the IV and DV). In examining the other data sets and comparing them to the data from Cast #1, it has been concluded that the salinity data for this cast is erroneous for the following reasons:

- The location of Cast #1 was in the vicinity of the AES HBGS outfall structure. All other CTD casts surrounded the outfall structure with casts being performed to the north (#2, #3, #4 and #8), inshore (#4, #5, #8 #9), offshore (#7, #11), and south (#6 and #10). The surface and near-surface salinity values for all other casts ranged from approximately 33.0 ppt to 33.8 ppt. This surrounding CTD data indicated that a layer of lower salinity water was not present in the area. The surface and near-surface salinity values measured during Cast #1 were isolated and localized in that one location;
- The range in salinities measured throughout the water column in the various locations during all other CTD casts ranged from a low of approximately 33.0 ppt to a high of approximately 34.9 ppt. Again, this surrounding CTD data indicated that lower salinity water was not present in the area. The surface and near-surface salinity values measured during Cast #1 were isolated and localized in that one location; and
- Salinity data from the IV and DV indicated that typical salinity ranges for the study period when the offshore CTD casts were performed were in the order of 33.3 ppt to 33.6 ppt. If water was being discharged from the outfall with a salinity of approximately 33.3 ppt and

lower salinity water was not present in the vicinity of the outfall (as detailed from the data presented above), it would not be possible for salinities to decrease to the low values seen in the Cast #1 data set (30 ppt) in the vicinity of the AES HBGS discharge location.

Based on the above, the salinity values from Cast #1 have not been included in any of the subsequent analyses and discussions as part of this report, but all other CTD data at all stations has been included.

Surface water samples were collected using a Niskin triggered below the water surface (but within the upper 1 m). Care was taken to ensure a near-surface sample.

Surfer (Win32) v. 6.04 Surface Mapping System was used to contour the data sets. Kriging gridding method was used (exact interpolation) with a 50 x 50 grid size. Data points were posted to highlight locations of data sets.

Surfer is a grid based contour program. Gridding is the process of using original data points (observations) in an XYZ data file to generate calculated data points on a regularly spaced grid (a grid [.GRD] file). Interpolation schemes estimate the value of the surface at locations where no original data exists, based on the known data values (observations). Surfer then used the grid to generate the contour map or surface plot.

Most of the gridding methods in Surfer use a weighted average interpolation algorithm (i.e. with all other factors being equal, the closer a data point is to a grid node, the more weight it carries in determining the Z value at a particular grid node). Kriging is one of the more flexible methods and is useful for gridding almost any type of data set. Kriging is the default gridding method because it generates the best overall interpretation of most data sets.

Exact interpolators honor data points exactly when the data point coincides with the grid node being interpolated. Even when using exact interpolators it is possible that the data is not honored exactly by the grid file if the data points do not coincide with the grid nodes. To increase the likelihood of the data points being honored, increase the number of grid lines in the X and Y direction (in this case a 50 x 50 grid size). This increases the chances that grid nodes coincide with the data points, thereby increasing the chance that the data point values are applied directly to the grid file.

4.5.1.4 CTD Data Obtained During the Offshore Dye Study

The horizontal ocean surface gradients of temperature and salinity at the AES HBGS outfall (**Figures 4-24** and **4-25**) clearly showed the influence of the thermal discharge as noted by the

warm water signature. The surface temperature signature from the AES HBGS outfall was shown to be approximately 2°C above the ambient surrounding seawater temperatures.

Operational parameters at the AES HBGS facility are included in **Table 4-2** for the entire period of study. A summary of this data collected specifically during the offshore dye study on August 20, 2002, is included in **Table 4-14** shown below.

Table 4-14 Summary of AES HBGS Operational Parameters on August 20, 2002

Sample Time	Intake Temperature (°F/°C)	Discharge Temperature (°F/°C)	Number of Discharge Pumps Operational	Outfall Flowrate (gpm)
00:30	65.0 / 18.3	85.0 / 29.4	5	220,000
03:45	65.0 / 18.3	85.0 / 29.4	5	220,000
07:15	65.0 / 18.3	85.0 / 29.4	5	220,000
09:25	65.0 / 18.3	85.0 / 29.4	5	220,000
13:10	65.0 / 18.3	85.0 / 29.4	5	220,000
15:30	65.0 / 18.3	85.0 / 29.4	5	220,000
18:45	65.0 / 18.3	85.0 / 29.4	5	220,000
21:50	65.0 / 18.3	85.0 / 29.4	5	220,000

Plant operational parameters for August 21, were similar with the exception of a lower discharge temperature of 80°F.

Figures 4-26 through **4-29** show data from the 12 CTD casts conducted as part of the offshore dye study. This data includes temperature, salinity, dissolved oxygen and pH, all as a function of depth. The locations of individual CTD cast have been shown previously on **Figure 4-24** and **4-25**. Cast #1 was in the vicinity of the AES HBGS outfall, while Cast #7 was in the vicinity of the intake. Casts #2, #3, #4 and #8 were collected to the northwest and inshore of the intake and discharge points. Casts #9 and #5 were collected to the northeast and inshore of the discharge, while Casts #6, #10 and #11 were collected to the southeast of the discharge point. Cast #12 was a single cast conducted the following day on August 21, 2002, in the vicinity of the AES HBGS intake structure.

Figure 4-26 presents the vertical temperature profiles for all 12 CTD casts collected as part of the offshore dye study. Data from Cast #1 (conducted near the AES HBGS discharge location)

showed the distinct temperature stratification, with a prominent thermocline at a depth of approximately 2 meters beneath the surface. The data showed that the warm water discharge from the outfall drives temperature stratification in the vicinity of the AES HBGS outfall. The maximum water column vertical temperature change occurred at the outfall and was approximately Δ 3.0 °C. Data from the other CTD casts collected in the area on August 20, 2002, showed less thermal stratification with vertical temperature changes ranging from approximately Δ 0.9 °C to Δ 1.7 °C.

Figure 4-27 presents the vertical salinity profiles for 11 of the 12 CTD casts collected as part of the offshore dye study. Data from Cast #1 has not been included for reasons discussed previously in the sections detailed above. Salinities in the study area on August 20, 2002, typically varied from a low of approximately 33 ppt. (at the surface during Cast #9) to a high of 34.5 ppt (at depth during Cast #4). The vertical salinity gradients were in the order of approximately Δ 0.73 ppt to Δ 1.47 ppt, which indicated well-mixed conditions.

Figure 4-28 presents the vertical dissolved oxygen profiles from all 12 of the CTD casts collected as part of the offshore dye study. The greatest depression in dissolved oxygen concentrations occurred in the vicinity of the AES HBGS discharge and had an approximate Δ of 0.46 mg/L with the range of dissolved oxygen concentrations varying from a low of 7.47 mg/L to a high of 7.93 mg/L. Elsewhere in the study area, the dissolved oxygen concentrations varied between 7.60 mg/L and 7.98 mg/L, with approximate Δs of 0.12 mg/L to 0.33 mg/L.

Figure 4-29 presents the vertical pH profiles from all 12 of the CTD casts collected as part of the offshore dye study. In all cases the pH of the water column varied between a low of 8.21 to a high of 8.25. Approximate vertical pH Δs varied from a low of 0.00 to a high of 0.03.

4.5.1.5 Indicator Bacteria Data During the Offshore Dye Study

During the offshore dye study on August 21, 2002, a total of six water samples were collected offshore near the surf zone of Huntington State Beach. These water samples were analyzed for the indicator bacteria total coliform, fecal coliform and Enterococcus. A summary of the data is presented (**Table 4-9)** and includes sample identification, time, latitude, longitude and bacterial concentration.

4.5.2 OFFSHORE DYE STUDY – PLUME IMAGES

During Tuesday, August 20, 2002, a series of five dye releases was performed. Rhodamine WT dye was injected into the discharge outfall of the AES HBGS at a known rate. Time-series, high-

resolution, digital photographs of the dye fields were then collected. Time-series, enhanced-color images have been presented for four of the dye releases. Because of the sun angle during the fifth dye release, useable images were not obtained. Summary information for each of the image series is detailed below as follows:

- Dye Release #1 **Plates 4-16a** and **4-16b** Images from 10:50 to 11:35;
- Dye Release #2 **Plates 4-17a** and **4-17b** Images from 12:15 to 12:45;
- Dye Release #3 **Plate 4-18** Images from 13:41 to 14:46;
- Dye Release #4 **Plates 4-19a**, **4-19b** and **4-19c** Images from 15:41 to 16:46.

The first and last images in each series are true-color images and have been included for comparison purposes with the enhanced-color images.

Attempts were made to calibrate the colors of the dye fields in the images to actual dye concentrations measured by the survey vessel on the ocean surface. Because of the sun angle and cloud cover during the morning and early afternoon dye releases, it was not possible to obtain accurate color calibrations for images as part of Dye Release #1 and #2. However, these calibrations were successfully completed for Dye Release #3 and Dye Release #4. As such, the images for the latter two dye releases also include a calibrated color bar located to the right of each image. This shows the variation in dye concentration as a function of the observed color.

The images shown on **Plates 4-16a** and **4-16b** are from Dye Release #1 which started at 10:35 and finished at 10:45. During the 10-minute dye-release period, Rhodamine WT was injected into the discharge from the HBGS at a rate of 260 ml/minute (resulting in a Rhodamine WT concentration at the end of the outfall of approximately 62.4 µg/L [ppb]).

The images shown on **Plates 4-17a** and **4-17b** are from Dye Release #2 which started at 12:01 and finished at 12:31. During the 30-minute dye-release period, Rhodamine WT was injected into the discharge from the HBGS at a rate of 300 ml/minute (resulting in a Rhodamine WT concentration at the end of the outfall of approximately 72.4 µg/L [ppb]).

The images shown on **Plate 4-18** are from Dye Release #3 which started at 13:30 and finished at 14:00. During the 30-minute dye-release period, Rhodamine WT was injected into the discharge from the HBGS at a rate of 300 ml/minute (resulting in a Rhodamine WT concentration at the end of the outfall of approximately 72.4 µg/L [ppb]).

The images shown on **Plates 4-19a**, **4-19b** and **4-19c** are from Dye Release #4 which started at 15:30 and finished at 15:56. During the 26-minute dye-release period, Rhodamine WT was

injected into the discharge from the HBGS at a rate of 300 ml/minute (resulting in a Rhodamine WT concentration at the end of the outfall of approximately 72.4 µg/L [ppb]).

Interpretation of these image series as they relate to dilution and shoreward transport is included in Section 5 of this report.

4.5.3 COMPUTER MODEL OF THERMAL PLUME

4.5.3.1 Model Outputs

Appendix F contains the computer printouts generated with the input variables described above for the August 2002 stratification only. May and September stratifications yielded nearly identical results. Accordingly, only the August 2002 detailed results have been included **(Appendix F).**

Discrete model runs are presented individually (**Appendix F**). The first section is a record of the input data (in metric units), and the near-field calculations (from the point of discharge to the ocean surface) are summarized in the second section. The final section details a tabular summary of the pollutant concentration and the total physical dilution (product of initial dilution and dilution due to dispersion) for each of the 4/3-power law and the constant eddy diffusion algorithms.

The concentration values in the final section are based on an initial concentration (see input variables) of Enterococcus at 100 cfu/100ml, facilitating separation of the dilution due to bacterial die off (100/C_t /Dilution) from the other components. The concentration approximates the AB 411 single-sample criteria for Enterococcus bacteria.

4.5.3.2 Initial Dilution

The model was initially run for three seasonal conditions (May, August and September). The model predicts that the plume surfaces quickly in all seasonal conditions with little appreciable difference in the predicted initial dilution values. As noted previously, this occurs because the discharge depth is minimal; the port is large and oriented vertically. Additionally, the momentum flux is large because of the thermal differences between the ambient receiving water and the thermal discharge. Therefore, the buoyant, passive plume has little opportunity to mix with the receiving water as it rises to the ocean surface, which results in a minimal reduction in concentration until the plume reaches the surface and additional dilution processes commence. The dilution values for the three stratifications investigated ranged from:

- 1.28:1 for the highest discharge rate (352,000 gpm or 22.21 m^3/s) in the slowest ambient current (0.05m/s); to
- 1.43:1 for the lowest discharge (220,000 gpm or 13.88m^3/s) in the fastest ambient current (0.15m/s).

The two discharge rates were selected on the basis of five operational pumps during the duration of the dye study, and a maximum of eight pumps potentially operational. The variation is so minimal, that for all practical purposes, there is no difference in these initial dilution values. In the subsequent predictions for water quality parameters, the lowest dilution values (largest flows in the slowest current) have been used to generate conservative predictions.

4.5.3.3 Dilution Due to Dispersion

Dilution due to dispersion is time related, which in turn is directly related to the distance between the discharge and the point of reference, and inversely proportional to the current speed. The shortest distance between the shoreline and the point of discharge is 1,200 feet (365 meters).

Pumps (#)	Discharge Rate (gpm)	Discharge Rate (m^3/s)	Current Speed (m/s) 0.05 Dilution Value	0.10 Dilution Value	0.15 Dilution Value
5	220,000	13.88	13.1	6.0	4.1
6	264,000	16.66	12.6	5.8	3.9
7	308,000	19.43	12.4	5.6	3.9
8	352,000	22.21	12.1	5.5	3.8

Predictions were based on current velocities of 0.05 m/s to 0.15 m/s. The products of the initial dilution and the reduction in concentration due to dispersion (representing total physical dilution) using the 4/3 eddy diffusion algorithms are summarized below:

Bacterial Disappearance

The rate of bacterial die-off can be predicted using Chick's Law (Velz, 1984), which indicates that bacteria in an unfavorable environment die at a constant rate (a given percentage of the remaining population will die during each successive time unit).

The values shown in the table are presented in a "dilution" form for ease of reference when comparing them to the initial and secondary dilution values. The dilution values are again represented as number of dilutions relative to one. The bacterial die-off dilution value is another way of representing a certain percentage of bacterial die-off. The inverse of the dilution value is the percentage of bacteria remaining. For example, a bacterial die-off dilution value of 2.2 means that 45% (1/2.2) of bacteria remain alive at that point after traveling from the outfall to the nearest shoreline contact point.

The variation in bacterial disappearance (expressed in terms of an additional dilution [#:1]), as a function of T90 and current speed, at the closest point of shoreline contact (1,200 feet [350 m]) is summarized below.

T90 (hours)	Shore Distance (m)	Current Speed (m/s)			
		0.05	0.10	0.15	0.20
4	350	3.06	1.75	1.45	1.32
8	350	1.75	1.32	1.21	1.15
16	350	1.53	1.24	1.15	1.11

Model Sensitivities

The model was found to be relatively insensitive to changes in the outfall discharge rate and ambient water column profiles. Hence, initial dilution values were relatively low and very consistent regardless of flow rate or seasonal condition. Secondary dilution values were strongly impacted by the selection of the far-field dispersion factor. The value recommended for open oceans was discarded in favor of the more conservative default value in the program. The model was also found to be very sensitive to bacterial die-off times (T90).

4.5.4 OFFSHORE DYE STUDY – DYE OCCURRENCE IN THE SURF ZONE

Table 4-8 and **Figure 4-30** presents a summary of all Rhodamine WT concentrations in water samples collected at the shoreline monitoring stations (6N, 7.5N, 9N, 10.5N, 12N, 13.5N and 15N) during the offshore dye study from 0930 on August 20 to 1530 on August 21, 2002. The figure shows Rhodamine WT dye concentrations by individual station as a function of time and as related to the tide height during the study period. Superimposed on each figure is the time

of occurrence of each dye release. Some general observations regarding the shoreline dye concentrations include the following:

- **Station 6N**. 82 samples were collected at this station. The concentration of Rhodamine WT dye varied from a low of 0.00 micrograms per liter (µg/L) to a high of 0.43 µg/L with a mean concentration of 0.07 µg/L.
- **Station 7.5N**. 55 samples were collected at this station. The concentration of Rhodamine WT dye varied from a low of 0.02 micrograms per liter (µg/L) to a high of 0.16 µg/L with a mean concentration of 0.06 µg/L.
- **Station 9N**. 80 samples were collected at this station. The concentration of Rhodamine WT dye varied from a low of 0.00 micrograms per liter (µg/L) to a high of 0.63 µg/L with a mean concentration of 0.08 µg/L.
- **Station 10.5N**. 55 samples were collected at this station. The concentration of Rhodamine WT dye varied from a low of 0.01 micrograms per liter (µg/L) to a high of 1.98 µg/L with a mean concentration of 0.17 µg/L.
- **Station 12N**. 82 samples were collected at this station. The concentration of Rhodamine WT dye varied from a low of 0.01 micrograms per liter (µg/L) to a high of 1.05 µg/L with a mean concentration of 0.18 µg/L.
- **Station 13.5N**. 54 samples were collected at this station. The concentration of Rhodamine WT dye varied from a low of 0.00 micrograms per liter (µg/L) to a high of 0.60 µg/L with a mean concentration of 0.14 µg/L.
- **Station 15N**. 55 samples were collected at this station. The concentration of Rhodamine WT dye varied from a low of 0.00 micrograms per liter (µg/L) to a high of 0.47 µg/L with a mean concentration of 0.12 µg/L.
- The highest concentrations of dye were observed at Stations 10.5N and 12N, and
- The first station that dye was detected at varied with deployment. For the first deployment, dye was first observed to contact the shoreline at Station 12N. Later in the day, station 10.5N was the first location where the dye was first observed.

Interpretation of this data as it relates to dilution of the AES HBGS thermal plume and potential bacterial concentrations along the shoreline is included in Section 5 of this report.

4.5.5 INTAKE AND OUTFALL ASSESSMENT

4.5.5.1 Mooring Observations and IV

In late summer, the water column is generally stratified. The mooring time series shows that the water column was stratified most of the time during August-October, 2002 **(Figure 4-31)**. Temperatures were generally cooler during the second deployment and the temperature difference between the surface and ten meters was somewhat less. During the first deployment there were two periods when the water temperatures were cooler, particularly in the lower part of the water column. Between September 1 and 8, 2002, temperatures routinely fell below 14°C. Near bottom temperatures also fell below 14°C during September 27 to 29 and October 14-to 16, 2002. Stratification was markedly reduced between August 27 and 29, 2002, but this was during a period when bottom temperatures were warmer.

The quality of the September data is somewhat poorer, showing some scatter in salinity. There was substantial drift in the conductivity sensor, which is only partially accounted for here. The temperatures entering the plant were cooler in early September **(Figure 4-32).**

4.5.5.2 Discharge Vault Observations

The effects of the power plant on the water drawn into the plant are shown in **Figure 4-32**. The time series from the sensor in the IV and the two sensors in the discharge vault are shown. The most obvious change that occurs is the warming of the water. The warming is phased through the day due to changes in plant operation. Warming of 10-15°C occurred during the second half of the day; around midnight the warming would only be about 2-3°C. It appears that there was a steady input of freshwater into the output stream as it is discharged. In the period after August 26, when the sensor was at a depth of about six meters (pressure six dbar), the discharge salinity as measured by the sensors in the discharge vault were consistently lower than the salinity in the IV. There is some question about the absolute salinity of the deeper sensor in the discharge vault particularly since it is lower than the salinity of the shallow sensor during the period of August 20 to 23, 2002. The pressure record for the deeper sensor in the discharge vaults shows that the sensor slipped deeper in the vault in two steps. The first step was on August 23, when the sensor slipped about one meter, and the second step occurred on August 25 when the sensor slipped another three meters and actually slipped into the discharge pipe itself.

The change in salinity during the first deployment between the intake and discharge vault (deep SB-MC) has been assessed **(Figure 4-33).** During the first 12 days the SB-MC was at mid-

depth **(Figure 4-32)** as described above, for the last six days the SB-MC was in the discharge pipe and therefore enabled a comparison of the change in salinity between intake and discharge water. During this six-day period water exiting the discharge pipe was more saline than water at the surface of the discharge vault and occasionally was at an equivalent salinity (August 28 to September 01, 2002). An assessment of the likely volume and sources of the freshwater input observed in the discharge vault has been discussed in **Section 5.6.3.2.**

The two discharge rates were selected on the basis of five operational pumps during the duration of the dye study, and a maximum of eight pumps potentially operational. The variation is so minimal, that for all practical purposes, there is no difference in these initial dilution values. In the subsequent predictions for water quality parameters, the lowest dilution values (largest flows in the slowest current) have been used to generate conservative predictions.

4.6 MICROBIAL SOURCE TRACKING

The microbial source tracking techniques used in this investigation, were able to differentiate the source of bacteria or virus as either human (positive) or non-human (negative).

4.6.1 BACTEROIDES/PREVOTELLA ANALYSIS

Of the *Bacteroides* and *Prevotella* bacterial source tracking samples collected (64), only two were confirmed as positive (human source) **(Table 4-10)**; one from the Intake Vault on 21 August at 9:40 PM (very near high tide), and the Discharge Vault on 22 August at 10:05 PM (also near high tide). Parallel measurements (not funded by this project) showed multiple positive results (human source) for this assay during the evening high tide of 21 August, 2002 at locations all along the beach, at the Santa Ana River, Talbert Marsh, 6N, 9N, 12 N, 15N, and 18N sites. This was not a period when the water at the intake mooring was particularly cool, i.e. when potential entrainment of subthermocline water would occur, bottom temperature was greater than 17°C.

4.6.2 ENTEROVIRUS ANALYSIS

Of the Enterovirus bacterial source tracking samples collected (59), only two tested positive (human source); in the intake vault at 3:10 on August 21, 2002; and the general purpose retention basin at 11:20 on August 22, 2002. When the GP sample was retested approximately one and a half months after the original test (sample stored frozen at -80°C), the result was negative. The inability to obtain a repeat positive indicates that the result was a false positive;

because of the highly fragile nature of RNA it is also possible that the RNA degraded during a repeat freeze-thaw cycle and storage.

4.7 QUALITY ASSURANCE/QUALITY CONTROL

QA/QC data is presented in **Tables 4-11, 4-12, and 4-13.**

5 DISCUSSION

5.1 GENERAL WATER QUALITY

Microbial water quality at Blackford's Ditch, the Boiler Fireside Wash and to a lesser extent the Boiler Sump Wash locations consistently exceeded AB 411 recreational water quality standards; microbial concentrations within the storm water sump and retention basin were lower than Blackford's Ditch, the boiler fireside wash and the boiler sump wash. The decrease in bacterial concentrations in the storm water sump and general purpose retention basin were probably due to dilution, die-off and predation of the bacteria.

The source of bacteria at the Blackford's Ditch sampling location was mixed and probably consisted principally of urban runoff (including irrigation runoff) and canine feces; the source of bacterial contamination of the Boiler Fireside Wash was most probably avian (pigeons-feathers observed in sample location). Bacterial concentrations at the intake vault and discharge vault were consistently low and only exceeded AB 411 21 times **(Table 5-1**), of these exceedances nine were observed in the intake vault and therefore represent an import of contamination from the ocean; ten were in DV_0 and are most probably from in-plant freshwater sources, and two were observed at DV_{10}. Of the two exceedances observed at DV_{10}, both occurred in conjunction with an exceedance in either the intake vault or at the surface of the discharge vault (the two exceedances at DV_{10} coincided with similar concentrations of Enterococcus at DV_0). In the absence of either contaminated intake water, or contaminated urban sources (offsite surface runoff); no exceedances appear to have been caused by the thermal load of the AES HBGS. A statistically significant difference between the IV and DV_0, DV_{10}, and DV_{30} sample locations was demonstrated. Total and fecal coliform concentrations in the IV were lower than DV_0 and DV_{10}, while Enterococcus concentrations were not significantly different **(Figure 4-12, Figure 4-13 and Figure 4-14)**. Bacterial concentrations for all three microbial analytes were lower at DV_{30} than for IV. The source of the bacterial contamination within the discharge vault is most likely a freshwater source (as confirmed by the correlation of salinity and bacterial concentrations at DV_0 and DV_{10}). The extent to which this source is able to impact the surf zone is dependant on several factors:

- Degree of stratification within the discharge vault;
- Number of circulation pumps operating;
- Tidal height and direction;
- Flow rate from the source locations (GP and BD initially);

- Cooling water temperature; and
- Ocean currents and dilution die-off effects.

Standard operational practices of the AES HBGS (excluding surface runoff treatment) do not significantly contribute to bacterial contamination of the cooling water. Considering the average retention time of water within the cooling system is only 20 minutes, and the maximum doubling time of *E. coli* under ideal conditions is 15 minutes (fecal coliform and Enterococcus doubling times are longer). It should be noted that the microbial conditions in the intake water are far removed from the ideal laboratory conditions that will result in an *E. coli* doubling time of 15 minutes, the exact doubling time of *E. coli* in the intake water is not known and was beyond the scope of this study, but it can be speculated to be in the order of hours as opposed to minutes (if growth occurs at all). The temperature rise between the IV and DV would not be anticipated to increase bacterial concentrations during standard plant operations. This assumption was confirmed by the absence of significant bacterial proliferation between IV and DV **(Figure 4-12, Figure 4-13 and Figure 4-14).** Similarly the heat treatments conducted during this investigation did not increase the concentration of bacteria within the cooling water **(Table 4-1 and 4-2).**

5.2 SANITARY SEWERS

During the 24-hour sanitary sewer dye test, no dye was detected in the discharge vault with an in-situ fluorometer, no dye was observed in either the DV or the GP. If the sewers had been connected to the discharge vault, not only would dye have been observed during the dye study, but periodic large concentrations of total and fecal coliform, and Enterococcus would have been observed in any one of three possible sample locations in the discharge vault, no such periodic fluctuations were observed during the 90 days of continuous daily sampling (and 14 days of sampling every three hours-**Table 4-1 and 4-2; Figure 4-23**).

5.3 BLACKFORD DITCH

The water quality of Blackford's Ditch was generally poor, with high concentrations of all three-indicator bacteria. The ditch remained at a constant depth of approximately two to three feet near the pump house gates at the western end of the ditch. During the summer, the only known source of freshwater into the ditch is from irrigation runoff. Irrigation runoff was observed entering the ditch at a rate of approximately one gpm for a period of between 30-240 minutes (at approximately 01:00 during the intensive two-week sampling period). The ditch may be receiving up to 120 gallons of irrigation runoff per day, and therefore a net flow of 120

gallons or more of BD water would flow into the DV in one day. The net flow of bacteria would be approximately 8.6 x 10^8 total coliform, 3.8 x 10^6 fecal coliform and 1.7 x 10^8 Enterococcus. If the BD is essentially flushed daily by the tides as indicated by the salinity concentrations observed in the ditch (**Figure 4-22**), then approximately half the total number of bacteria would be anticipated to be flushed out of the ditch with each tide. During standard plant operations this would equate to approximately 4 x 10^8 total coliform being discharged into 1.1 x 10^9 100ml aliquots of water (less than 0.1 total coliform per aliquot). It should be noted that only surface flow into the ditch was quantified, some irrigation flow may percolate into the ground and some of the irrigation water may be lost to evaporation and evapotranspiration. Discharge at the flapper valve adjacent to the discharge vault was difficult to determine due to high water levels.

Two of the three transects conducted on the ditch demonstrated a salinity gradient with distance from the pump house. The discharge vault has been considered as a potential route for seawater to back up into the ditch at high tides, the salinity concentrations observed in this study (up to 23.2 ppt on August 16, 2002) were consistent with such a mechanism. Salinity concentrations in Blackford's Ditch remained consistently high in comparison to freshwater throughout the study. During the storm event of November 9, 2002, the pump was activated (float-switch mechanism), and water from the ditch was discharged into the discharge vault. Salinity was less than five ppt on the subsequent transect conducted on November 26, 2002. The bacterial flux during a storm event would be significantly higher than during dry weather, resulting in an increase of 10 to 20-fold bacterial loading.

Throughout the duration of the study, the water in the ditch had a vivid green color, which intensified during September and October. The source of the discoloration of the water was not known, but two possibilities were:

- Periodic disposal of automotive coolant; and
- Corrosion inhibitor from the AES HBGS.

Samples were collected and analyzed for nitrite and boron (compounds present in the corrosion inhibitor) on October 14, 2002. Elevated concentrations of boron were observed **(Table 4-5)**, the minimum observed concentration (60.2 mg/L) was approximately 13 times greater than ocean water (4.6 mg/L-USGS, 1992). Elevated concentrations of nitrite were also observed, and could be accounted for from other sources including urban runoff. A second set of samples was collected on November 26, 2002, after the storm event, after the ditch had received a significant freshwater input and had been partially pumped. Samples were collected for boron and nitrite as previously sampled, and tolytriazole (another ingredient in the corrosion inhibitor). Both

boron and nitrite concentrations were significantly lower in November than October; however, tolytriazole was detected in three samples above 0.2 mg/L. The presence of tolytriazole and the previously observed elevated concentrations of boron would suggest that corrosion inhibitor may be leaking into Blackford's Ditch from the AES HBGS.

5.4 STORM WATER SAMPLING

The storm event of November 8 to November 12, 2002 provided an opportunity to assess the impact of storm water on water quality. Samples were collected at the intake and discharge vaults, retention basin and three offsite storm drains (including Blackford's Ditch). Every sample collected exceeded AB 411 for *Enterococcus*, and most failed AB 411 standards on all three analytes **(Table 4-7)**. Ocean water coming into the AES HBGS exceeded AB 411 standards (*Enterococcus*) by two-fold (i.e. greater than 208 cfu/100ml). The highest concentrations of bacteria were detected in the storm drain locations. Canine feces were present in large quantities by Blackford's Ditch and the storm drain by PCH. Hydrocarbons (visible as a sheen) were documented entering Blackford's Ditch and were traced back to the automotive facility on Edison Avenue. No storm water run off was observed leaving the City of Huntington Beach Yard at the end of Edison Avenue.

During the storm event the AES HBGS was performing a heat-treatment process, consequently it is not possible to conclude what additional impact the storm drains had on the already impaired ocean water that was being drawn into the AES HBGS that day. However, it is clear that any storm-related discharge from the storm drains into the discharge vault will impair water quality in the vault itself, and potentially the surf zone.

5.5 AMMONIA

The concentration of ammonia in ocean water has been routinely used as an indicator of wastewater contamination from a variety of sources including urban run-off and submerged ocean outfalls. Initially, ammonia concentrations were to have been assessed in this study at the intake mooring and the intake vault. However, due to equipment difficulties this was not possible, consequently ammonia samples were routinely collected at the IV and DV locations and analyzed by Environmental Protection Agency Methods for Chemical Analysis of Water and Wastes (MCAWW)-EPA/600/4-79-020 (350.1). The concentrations were reported in milligrams per Liter (mg/L) as NH_3.

The intake and discharge vault sample locations (IV, DV0, DV10 and DV30) were sampled routinely, occasionally other sample locations within the AES HBGS (BFW, SWS, GP and BFW)

and outside the AES HBGS (BD) were also sampled. Equipment and Field Blank samples were collected for ammonia along with other parameters. All sample blanks (using de-ionized water) were below detection limits for ammonia (0.100 mg/L), and in-plant samples ranged from 0.22 mg/L (SWS August 23, 2002) to 24.2 mg/L (BFW August 23, 2002). Considerable variation in in-plant ammonia concentrations was anticipated because of routine flushing, and high values in the BFW location were consistent with the presence of bird feathers in the sample location.

The concentrations of ammonia in the coastal ocean are typically less than 0.01 mg/L (0.5 μM) with slightly higher values in the bottom boundary layer. In the sewage plume from the OCSD outfall the maximum concentrations are typically approximately 0.06 mg/L (3 μM). Concentrations in the range of 7-9 μM have been observed on at least one sample event (MEC, 2001), directly over the OCSD outfall; corresponding to 0.12 to 0.15 mg/L, slightly above the detection limit of the methodology used in this study (0.100 mg/L).

Table 5-3 Range of Ammonia Concentrations in IV and DV Samples

Sample Location	Minimum Ammonia Concentration (mg/L)	Maximum Ammonia Concentration (mg/L)	Mean Ammonia Concentration (mg/L)
IV	1.36	3.90	2.2
DV-0	1.20	3.65	1.91
DV-10	1.00	3.55	1.71
DV-30	1.05	2.80	1.85

None of the ocean water samples had ammonia concentrations of less than 1.0 mg/L. The ammonia values observed in the cooling water samples (IV and DV) were approximately 200-fold higher than would be routinely observed in ocean water (**Table 5-3**). The average concentrations for the four sample locations were between 1.71 and 2.2 mg/L.

As an example, an entrained OCSD outfall plume with minimal dilution would produce ammonia concentrations of only 0.12 to 0.15 mg/L (only 10% of the total ammonia concentration observed, and assuming typical ammonia concentrations in the OCSD plume, which were not known for that sample period). The proximity of the OCSD plume during the ammonia sampling was not known, and other sources of ammonia such as Santa Ana River, San Gabriel River and Los Angeles River exist in this region. Although ammonia has been historically used in powerplant boiler systems, a significant and sustained input would be required to produce the concentrations of ammonia observed in this study, if the data obtained is to be considered reliable.

5.5.1 RELIABILITY OF EPA AMMONIA METHOD EPA/600/4-79-020

Ammonia was analyzed using EPA method 350.1 for all samples collected, this method covers the determination of ammonia in drinking, surface, and saline waters, domestic and industrial wastes in the range of 0.01 to 2.0 mg/L NH_3 as N. The detection limit as reported by the analytical laboratory was ten-fold higher at 0.1 mg/L. Interferences in the analytical methodology can be caused by:

- Calcium and Magnesium ions can cause errors by precipitating during analysis;
- Turbidity; and
- Any color absorbing in the range of the indophenol blue complex.

The accuracy and sensitivity of this methodology has been questioned when applied to ammonia in ambient water (Griffith, 2003; Washburn 2003), particularly when applied with an elevated detection limit. The consistently high values observed for ammonia during this study support the suggestion that accuracy and sensitivity of the methodology may have been responsible, as it is highly unlikely that between August 19, and October 15, 2003 ammonia concentrations in the IV and DV sample locations were consistently above 1.0 mg/L.

5.5.2 BIOFOULING

Bioufouling of powerplant facilities by mussels and other macro-invertebrates is a well-established phenomenon. Bivalves and other macro-invertebrates are also known to secrete nitrogenous waste, including ammonia. As a second possible explanation for the elevated ammonia concentrations, biofouling has been considered, generally, whilst bivalve populations resident in the intake conveyances at the AES HBGS may contribute ammonia to the intake water, a density of approximately 500,000 bivalves /m^2 would be required to be solely responsible for the ammonia concentrations observed. It is not clear whether such a population is present even on a transient basis, and with the routine use of heat-treatment cycles by the AES HBGS to reduce biofouling, it is unlikely that such a high population would have been maintained throughout the duration of ammonia sampling (August 19 to October 10, 2002) which coincided with several heat treatments (**Table 4-1 and Table 4-2**).

5.5.3 AMMONIA SUMMARY

Irrespective of the influence of bivalves on elevated ammonia concentrations observed in this study, the ammonia data obtained are not useful as planned in determining the presence of a wastewater plume nearshore. Further analysis of this data is unlikely to change this conclusion,

and consequently the high ammonia values obtained can not be used in an assessment of the sub-thermocline hypothesis.

5.6 NEARSHORE OCEAN ENVIRONMENT

5.6.1 INTAKE ASSESSMENT

5.6.1.1 Observations

5.6.1.2 Mooring Observations and Intake Vault

The temperature time series from the intake vault was overlaid on the temperature time series from the intake mooring to compare the two time series (**Figure 5-2**). Temperature in the intake vault appeared to be consistent with temperatures from mid-depth on the mooring, suggesting that the cooling water drawn into the power plant was coming from mid-depth in the water column.

5.6.1.3 Evaluation of the Sub-Thermocline Mechanism

Grant et al. (2000) hypothesized that large cross-shelf excursions of the pycnocline due to internal tides and swash could transport the OCSD effluent plume cross-shelf into shallow water where it could be drawn into the AES HBGS cooling water intake stream. To assess this hypothesis the location of OCSD wastewater plume would have to be known. Identifying the location of the OCSD wastewater plume was beyond the scope of this investigation, and consequently conclusions relating specifically to the Subthermocline Entrainment Hypothesis can not be drawn. However, the potential for entrainment of the sub-thermocline layer in which the OCSD wastewater plume may be found, was assessed. Consequently with the data set obtained, the occurrence of the sub-thermocline mechanism was assessed during the observational period in August-October 2002 and compared with data from August 2002.

In order to evaluate whether subthermocline water is entrained into the AES HBGS cooling water, the temperature-salinity (T-S) scatter plot for the intake vault was overlaid on an example T-S data set from OCSD water quality observations from August 2002 in the region offshore between the 20 and 100 meter isobaths (**Figure 5-3**). Within the plot, red symbols indicate the T-S from the first intake vault deployment between August 13 and 31, 2002. The T-S from offshore is indicated with the blue plus symbols. The wedge of points where salinity was below 33.45 psu and temperature less than 13°C indicates the portion of the observations where the OCSD plume is located. In general, the presence of an outfall plume in a stratified water column appears as a wedge of points where the salinity is less than the ambient T-S. The

ambient T-S regions for the upper and lower layers are indicated in **Figure 5-3**. During August 2002, the top of the wedge of T-S indicating the plume was about 13°C. The salinity data from the first intake vault deployment in August looks qualitatively quite good (red plus symbols in **Figure 5-3**). During this period the temperature and salinity of the water entering the intake vault were well above the values measured for the effluent plume in August 2002. Based on this comparison, no subthermocline water at the level of the OCSD effluent plume could be drawn into the intake during the August period. However, some portions of the August T-S data show warmer lower salinity water coming into the intake vault. This portion of the T-S distribution is typical of the T-S characteristics of water from the river plume of the Los Angeles and San Gabriel Rivers. This plume often extends downcoast from Long Beach beyond the Santa Ana River mouth, can extend offshore a distance of 3-5 km, and typically has a thickness of 5 meters or less (Noble et al., 2003).

August 16-31

During the period of August 16-31, 2002, water temperatures were generally above 15°C at the 10 meter instrument on the mooring and slightly warmer within the intake vault. During one brief period on August 26 between 0600 and 0630 PDT temperatures at the intake mooring decreased almost to 14°C (**Figure 5-4**, Event 1, top panel). Within the intake vault the minimum temperatures were only approximately 15.6°C. The time lag between these two minima was about 45 minutes. The warmer temperature and higher salinity within the intake vault for this event indicates that the water within the intake vault was from mid water column and may have been a mixture of water between the surface and bottom.

September 1-24

During the period of September 1-24, water temperatures were cooler at the 10-meter depth of the intake mooring than during August. In the T-S plot the temperatures fall to nearly 13.5°C during several periods that are shown in **Figure 5-4** (top panel). This is still warmer than the water at the top of the sub-thermocline water where the plume was observed offshore. The three events when this occurs are indicated by the numbers 2, 3 and 4. The specific periods are listed in Table 2. The third period spans 3.5 days with the cooling events occurring at approximately 10-12 hour intervals, similar to the semidiurnal lunar tidal period (12.42 hours). Two exceedances were observed during the periods when intake temperatures were cooler-one on September 2 in the intake vault (Enterococcus) and one on September 7 for fecal coliforms at 9N.

Table 5-4. Summary of Intake Mooring Cool Water Events. (Temperature at the 10m instrument on the intake mooring was less than 14°C.)

Event	Period	Total coliform/ fecal coliform / Enterococcus (cfu/100ml) at 9N	Total coliform/ fecal coliform / Enterococcus (cfu/100ml) at IV
Event 1	August 26, 2002 0608	20/20/80	40/11/3
Event 2	September 1, 0824 – September 2, 2358	130/130/68 40/20/34	1/<1/<1 1/<1/**120**
Event 3	September 4, 1754 – September 8, 1000	80/80/52 20/20/38 500/**500**/70 20/<20/12	27/9/5 3/<1/<1 8/3/1 2/1/1 13/2/5
Event 4	September 15, 0722 – September 15, 1424	20/20<2 20/<20/8	3/2/<1

AB 411 Exceedances are shown in bold.

Based on the data presented above, there is no indication of sub-thermocline water at the density level analogous to the effluent plume intruding shoreward to near the AES intake. In August the intake water was both warmer and saltier than would be expected for sub-thermocline water. During September, slightly cooler water was observed at the 10 meter depth of the mooring near the intake but it was still warmer than the water where the OCSD plume was observed (**Figure 5-5**). Water temperature at the ten meter depth dropped as low as 13.5°C on three separate occasions (September 2, 6 and 8, 2002.). When the water temperature dropped to 13.5 °C, it was only for short periods of time (less than 4 hours on September 2; for an hour or two on September 8, 2002; and on September 6, 2002 the water temperature reached 13.5°C for a few minutes). The corresponding temperature in the intake vault did not drop below 14°C because the water entrained into the intake comes from above the sea bottom and is probably a mixture of water from different depths near the mooring. When the internal swash events occur, the coastal currents are typically downcoast such that the effluent plume from the OCSD outfall would be advected downcoast toward Newport (Noble et al., 2003). During this period the OCSD has started chlorinating (and dechlorinating) discharged effluent, although not at maximal potential in August.

In order for the 'swash' water to contain effluent, currents would need to be near zero, or have reversed in direction from downcoast to up coast coincident with a cross-shelf swash that would bring the sub-thermocline water shoreward.

5.6.2 OFFSHORE DYE STUDY – PLUME IMAGES

Time-series images from Dye Release #1 as shown on **Plates 4-16a** and **4-16b** show the dye surfacing directly over the AES HBGS outfall and then spreading radially in all directions out from the center of the discharge. Dye Release #1 occurred approximately one third of the way through an ebb tide. The influence of ambient ocean currents on the dye plume in this series of images is not clearly evident, compared to later dye releases. As discussed previously, because of the low sun angle and cloud cover during this dye release, it was not possible to accurately calibrate the image colors with dye measurements obtained by the survey vessel on the ocean surface.

Time-series images from Dye Release #2 as shown on **Plates 4-17a** and **4-17b** show the dye surfacing directly over the AES HBGS outfall and then spreading radially in all directions out from the center of the discharge. Dye Release #2 occurred approximately half of the way through an ebb tide. The influence of ambient ocean currents on the dye plume is seen in later images (085+) as the upcoast edge of the dye plume (on the left-hand-side of the images) is being flattened by the ambient currents flowing down coast (from left to right across the images). The hotspots seen in some of the images near the edge of the surf zone (069, 070, 072, 074, etc.) are artifacts from the color enhancement process (color specularity in the high energy wave environment of the surf zone) and are not occurrences of dye at the apparent concentrations observed in the center of an actual dye release. Again, as discussed previously, because of the low sun angle and cloud cover during this dye release, it was not possible to accurately calibrate the image colors with dye measurements obtained by the survey vessel on the ocean surface.

Time-series images from Dye Release #3 as shown on **Plate 4-18,** show the dye surfacing directly over the AES HBGS outfall and then spreading radially in all directions out from the center of the discharge. Dye Release #3 occurred just before slack water of an ebb tide. The images from Dye Release #3 were color-calibrated to accurately display the dye concentrations in the plume on the ocean surface as a function of enhanced color. The calibration was done using the imagery data and actual dye concentrations that were measured by the survey vessel on the ocean surface. The highest dye concentrations observed were directly over the center of the discharge and were in the order of 12 μg/L (parts per billion [ppb]). As the outfall was discharging dye with a maximum concentration of approximately 72 ppb, the measurement of 12 ppb directly over the outfall at the surface corresponds to an initial dilution of approximately 6 to 1. The radial spread of the dyed plume on the ocean surface resulted in additional mixing of the plume with ambient seawater and additional dilution of the initial dye slug. The

thickness of this surface plume layer was typically 0.5 to 1 meter. Dye concentrations decreased from approximately 12 ppb at the center to near 3 ppb within a radial distance of approximately 600 feet (183 meters), the approximate distance where the edge of the plume started to interact with the surf in the surf zone. With an initial dilution of 6 to 1, followed by additional dispersion resulting in another 4 to 1 dilution, the total dilution of the dye plume as it approached the edge of the surf zone was approximately 24 to 1. These values become important in later discussions regarding dye concentrations measured along the shoreline after the dye plume has passed through the surf zone. The remnants of the dye plume from Dye Release #2 can be seen in several of the images for Dye Release #3. The dye plume remnants are shown as a slightly curved, and smeared light blue line upcoast from the outfall (near the bottom left-hand corner of the images). The influence of ambient currents on the dye plume can clearly be seen on some of the later images (136, 141, 144, 149, etc.).

Plate 5-1 shows the inferred current pattern during the offshore dye study. The currents driven by the thermal discharge extend radially in all directions from the center of the discharge. Outside the surf zone, the ambient currents are inferred to be parallel to the coast and moving from the west-northwest to the east-southeast (left to right across the images). Within the surf zone, the currents, driven by waves, are flowing from the east-southeast to the west-northwest (right to left across the images). The effects of the ambient currents outside the surf zone appear to flatten the upcoast edge of the dye plume and concentrate dye on that face where the ambient currents converge with the discharge-generated radial currents as shown in the descriptive figure. In particular locations, this convergence causes the flattened plume to break through into the surf zone and where it is entrained and driven along the shore by wave driven flows (currents).

Time-series images from Dye Release #4 as shown on **Plates 4-19a, 4-19b** and **4-19c** show the dye surfacing directly over the AES HBGS outfall and then spreading radially in all directions out from the center of the discharge. Dye Release #4 occurred after the start of a flood tide. The images from Dye Release #4 were also color-calibrated to accurately display the dye concentrations in the plume on the ocean surface as a function of enhanced color. As with the previous dye release, the highest dye concentrations observed were directly over the center of the discharge and were in the order of 12 ppb. Again, as the outfall was discharging dye with a maximum concentration of approximately 72 ppb, the measurement of 12 ppb directly over the outfall at the surface corresponds to an initial dilution of approximately 6 to 1. Dye concentrations decreased from approximately 12 ppb at the center to near 3 ppb within a radial distance of approximately 600 feet (183 meters). With an initial dilution of 6 to 1, followed by additional dispersion resulting in another 4 to 1 dilution, the total dilution of the dye plume as

it approached the edge of the surf zone is approximately 24 to 1. The effects of the ambient currents outside the surf zone on the dye plume are clearly evident in several of the images (211, 212, 213, 214, 215, etc.). Dye appears to be concentrated in a curved band along the upcoast side of the dye plume (left side of the images). As with images from Dye Release #3, the dye plume appears to break through into the surf zone on the upcoast side. Also seen in the later images are the remnants from Dye Release #3 shown as the light-blue patches along the bottoms of the images.

The two general mechanisms for dye to enter the surf zone are via direct radial transport out from the center of the discharge, and from a convergent front that forms between the radial flow from the discharge and the ambient alongshore coastal flow that eventually flows in towards shore, crosses the surf zone and enters the wave-driven zone along the beach. During the offshore dye study (Tuesday, August 20, 2002) this front formed on the upcoast side of the discharge plume with breakthrough into the surf zone occurring north of Station 9N. It is likely, that with the ambient current direction reversed, that the front would form on the downcoast side of the discharge plume and the breakthrough into the surf zone would also occur south of 9N.

5.6.3 SALINITY DIFFERENCES

5.6.3.1 Differences in CTD Data Sets Between Instruments

The CTD data obtained by Komex during the offshore dye study on August 20 and 21, 2002, varied from the other CTD data sets obtained from USC offshore moorings MO1 and MO3, as well as the data obtained from CTD instruments installed in the IV and the DV of the AES HBGS. The two principal differences were as follows:

- In comparing temperature data from the USC offshore mooring MO1 and the nearest Komex offshore CTD cast (Cast #11), the Komex temperatures are approximately 0.9 °C lower than those recorded at the mooring; and
- The salinity data (as calculated from temperature, conductivity and pressure data) in the vicinity of the AES HBGS outfall, as measured during CTD Cast #1, showed very low surface and near-surface salinity values (in the order of approximately 3 to 4 parts per thousand [ppt] lower than the surrounding ambient salinities and the salinities measured in the IV and DV).

Minor differences in temperature and salinity data may occur because of equipment sensor differences related to the makes and models of the various CTD instruments. For example, the

Komex offshore CTD data was gathered using an YSI 6600, whereas the IV and DV CTD data was collected using SB-MC. The sections below discuss rectification of the temperature data and the anomalously low salinities measured during offshore CTD Cast #1.

Rectification of offshore CTD Cast temperature Data

As noted above, the Komex CTD cast temperature data appeared to be approximately 0.9 °C lower than temperature data recorded at USC offshore moorings. Rectification of the temperature data set for the Komex CTD data was performed by applying a linear temperature offset to the Komex temperature data. The temperature difference noted above was added to the Komex data, and this adjusted temperature data was used to recalculate salinities using the Practical Salinity Scale (PSS). The adjusted temperature data combined with the original conductivity data and the original pressure data was input to the PSS algorithm to generate adjusted salinities. An example of the original and adjusted temperatures and salinities compared to the USC data is detailed **(Table 5-5)**.

Table 5-5 Original and Adjusted CTD Temperatures

Description	T (°C)	S (ppt)
Komex (Original Data)	20.930	34.530
USC Mooring Data	21.814	33.633
Delta	**-0.884**	**0.897**
Komex (Adjusted Data)	21.814	33.611
USC Mooring Data	21.814	33.633
Delta	**0.000**	**-0.022**

Surface Salinity Data From Cast #1

The salinity data measured during CTD Cast #1, in the vicinity of the AES HBGS outfall, showed very low surface and near-surface salinity values (approximately 3 to 4 parts per thousand [ppt] lower than the surrounding ambient salinities and the salinities measured in the IV and DV). In examining the other data sets and comparing them to the data from Cast #1, it has been concluded that the salinity data for this cast is erroneous for the following reasons:

- The location of Cast #1 was in the vicinity of the AES HBGS outfall. All other CTD casts surrounded the outfall structure with casts being performed to the north (#2, #3, #4 and #8), inshore (#4, #5, #8 #9), offshore (#7, #11), and south (#6 and #10). The surface and near-surface salinity values for all other casts ranged from approximately 33.0 ppt to 33.8 ppt.

This surrounding CTD data indicated that a layer of lower salinity water was not present in the area. The surface and near-surface salinity values measured during Cast #1 were isolated and localized in that one location;

- The range in salinities measured throughout the water column in the various locations during all other CTD casts ranged from a low of approximately 33.0 ppt to a high of approximately 34.9 ppt. Again, this surrounding CTD data indicated that lower salinity water was not present in the area. The surface and near-surface salinity values measured during Cast #1 were isolated and localized in that one location; and
- Salinity data from the IV and DV indicated that typical salinity ranges for the study period when the offshore CTD casts were performed were in the order of 33.3 ppt to 33.6 ppt. If water was being discharged from the outfall with a salinity of approximately 33.3 ppt and lower salinity water was not present in the vicinity of the outfall (as detailed from the data presented above), it would not be possible for salinities to decrease to the low values seen in the Cast #1 data set (30 ppt) in the vicinity of the AES HBGS outfall.

Based on the above, the salinity values from Cast #1 have not been included in any of the subsequent analyses and discussions as part of this report, but all other CTD data at all stations has been included.

5.6.3.2 Salinity Variations between Intake and Discharge Vaults

Salinity in the intake and discharge vaults was assessed with a Horiba U-10 meter, three SB-MC and samples analyzed by independent laboratories. The reliability of the Horiba data has already been assessed (**Section 4.1.2**) Salinity data from the IV and DV show differences in salinity **(Table 5-6).** SB-MC data was not included in **Table 5-6** but used for the assessment of salinity stratification and freshwater input. The Horiba data (field salinity) has not been used for the assessment of salinity stratification and freshwater inputs because of questionable reliability.

Two sets of SB-MC deployments occurred. In the first deployment between August 13 and September 1, 2002, one SB-MC was placed in the intake vault and two in the discharge vault. The depths in the discharge vault were initially set at the surface (mean low water level), and at ten feet deeper. However during the first deployment, the deeper SB-MC in the discharge vault slipped to a lower depth on two occasions (**Figure 4-32**); and for the last seven days (August 25 to September 1, 2003) was at a depth of approximately 20 feet below water surface (2 dbar in pressure to 6 dbar). The second deployment occurred between September and October 2002, in

similar locations, however the deeper SB-MC was not recovered and a salinity comparison for this period with the intake vault was not possible.

Table 5-6 Comparison of Salinity Values for the Intake and Discharge Vaults

Methodology	Minimum (ppt)	Maximum (ppt)	n	Std. Dev.	Mean (ppt)
Intake Vault					
Horiba U-10	30.30	35.10	78	1.16	**33.076**
Laboratory	31.000	34.400	26	0.907	**33.170**
Discharge Vault (DV30)					
Horiba U-10	30.10	34.10	71	1.32	**32.655**
Laboratory	30.799	36.500	24	1.157	**33.238**

During the first deployment, the deeper SB-MC salinity compared with intake salinity values once it had fallen to a depth of approximately 20 feet below water surface, (August 25, 2002 onwards). Prior to August 25, 2002, salinity in both discharge vault SB-MC was significantly lower than the intake vault SB-MC and therefore did not represent the salinity in the main discharge flow. Consequently, only the values from the period of August 25, 2002 to September 1, 2002 for the intake vault and deep SB-MC were used to assess any potential salinity variation, in conjunction with laboratory samples collected at a depth of 30 feet in the discharge vault sampled through the stainless steel well screen.

Between August 25 and September 1, 2002 salinity in the main discharge flow varied from –0.005 to +0.003 by comparison with the intake vault salinity. Using a range of –0.002ppt to –0.004ppt as the approximate salinity depression between intake and discharge vault, the volume of fresh water is calculated as 0.0056% to 0.011% of the total volume of water or 88, 066 to 44,033 gallons per day (mgd). In other units, that volume is approximately 30 to 60 gallons per minute (gpm) or 0.5 to 1.0 gallons per second (gps).

Known "fresh" water inputs to the DV system include the following:

- Flow from the Retention Basin; and
- Offsite flow from Blackford's Ditch, the wildlife car park and Newland Street.

Data obtained from the AES HBGS shows that flows from the Retention Basin averaged approximately 100,000 gpd during the month of August 2002. The maximum daily flow for that

same month from the Retention Basin was approximately 230,000 gpd. During dry weather input to the BD would be limited to irrigation run-off and other unspecified urban run-off flows. The Newland Street drain appeared dry throughout most of the sample period; the maximal range of freshwater from BD and the wildlife car park would be between 150-500 gallons (based in part on observations of irrigation run-off during the intensive sampling period). Specific flow volume data for offsite sources is not available.

Laboratory analyzed salinity samples reported an average higher salinity at DV-30 (33.238) than in the intake vault (33.170), the difference in means was 0.068 and standard deviation of each sample mean was 1.157 ppt (DV-30) and 0.907 ppt (intake vault). The difference in laboratory salinity means is not significant (n=24). Considering the insignificant variation between laboratory salinity assessment, and the small variation observed by the SB-MC, it would appear that a combination of the two known freshwater inputs would account for the estimated volume of freshwater detected by the SB-MC data.

5.6.4 OFFSHORE DYE STUDY – DYE OCCURRENCE IN THE SURF ZONE

As detailed previously, **Figure 4-30** presents a summary of all Rhodamine WT concentrations in water samples collected at the shoreline monitoring stations (6N, 7.5N, 9N, 10.5N, 12N, 13.5N and 15N) during the offshore dye study from 0930 on August 20 to 1530 on August 21, 2002. The figure shows Rhodamine WT dye concentrations by individual station as a function of time and as related to the tide height during the study period. Superimposed on each figure is the time of occurrence of each dye release. 462 individual shoreline samples were collected and analyzed during the study period. Dye was first observed to contact the shoreline at Station 12N during the first releases which correlates with the observed "concentration front" that formed on the upcoast side of the dye releases and then later broke through into the surf zone north of Station 9N. The highest concentrations of dye were observed at Stations 10.5N and 12N (2 ppb and 1 ppb, respectively), again indicative of where the "concentration front" broke through the surf line and approached the beach. When the maximum of 2ppb occurred at 10.5N, the dye from that release was entering the surf zone between 9N and 10.5N. Thus, the entry point of the discharge plume varies as a function of the interaction between radial flow from the discharge plume, the ambient alongshore currents and the shear across the surf zone.

In examining the dye concentration peaks in the data prior to the time of the first dye release, a maximum background dye concentration of approximately 0.15 ppb has been assumed. Any dye concentration in the 462 individual samples lower than the threshold value, have been filtered out of the data and not included for analyses. On the basis of examining the filtered data set, **Figure 5-6** presents a histogram of dye concentrations greater than 0.15 ppb. The

majority of the data are between concentration ranges of 0.20 ppb to 0.50 ppb, with a small number (8) of individual sample concentrations greater than that value (0.55, 0.65, 0.7, 0.95, 1.10 and 2.00 ppb).

On the basis of the observed dye concentrations (greater than the threshold value of 0.15 ppb, a dye dilution histogram has been prepared (**Figure 5-7)**. The average dye concentration that was discharged from the AES HBGS outfall was approximately 72 ppb (for Dye Releases #2, #3, #4 and #5). The dye dilution values at the shoreline monitoring stations were calculated by dividing the dye concentration being discharged from the outfall by the dye concentrations measured at a particular time and location along the beach. For example, if the outfall dye concentration was 72 ppb and a concentration of 0.35 ppb was measured on the beach, the dilution of dye at that point is 72 ppb divided by 0.35 ppb, which equals a dilution of 206 to 1. The highest measured dye concentrations along the beach correspond to the lowest dye dilution values, and conversely, the lowest measured dye concentrations along the beach correspond to the highest dye dilution values. The lowest calculated dye dilution value was found to be approximately 36 to 1 (based on the highest observed dye concentration along the beach of 2 ppb). The average dilution value calculated based on dye concentration measurements along the beach was found to be approximately 277 to 1, with the maximum dye dilution value being approximately 462 to 1.

Comparisons between pre- and post-surf zone dilution values can be made. In the vicinity of the AES HBGS outfall, initial dilution values of dye at the center of the dye plume were found to be approximately 6 to 1. An additional dilution of 4 to 1 was observed extending radially from the center of the dye plume to the edge of the plume near the boundary of the surf zone approximately 600 feet (183 meters) inshore. The total dilution of the dye plume along that boundary was estimated at approximately 24 to 1.

Because significantly lower dye concentrations (and calculated significantly higher dye dilution values) were measured on the shore at various monitoring locations, the additional dilution can be attributed to mixing in the surf zone. For example, 72 ppb of dye being discharged from the outfall becomes 12 ppb at the ocean surface over the outfall as it mixes 6 to 1 with ambient seawater on its rise to the surface. Then that 12 ppb of dye at the center becomes 3 ppb out at the edge of the dye plume just outside the surf zone as it moves and mixes 4 to 1 with ambient seawater. Once inside the surf zone, the wave energy serves to further mix that 3 ppb dye until measured at one of the shoreline monitoring stations at a concentration that ranged from 0.15 ppb to 2 ppb. That additional decrease in dye concentration (and the corresponding increase in dye dilution value) was the result of surf zone mixing. On the basis of the measured dye

concentrations at the shoreline monitoring stations, the surf zone mixing results in a range of additional dilution values that extend from a low of 1.5 to 1, to a high of 19 to 1.

However, because the duration of the dye releases were relatively short (each less than approximately 30 minutes), the measured dye concentrations are less than those that would have been measured had the dye release been continuous over a long period of time. Hence the calculated dilution values are higher than those that would have been determined on the basis of a continuous, long-term dye release. The maximum dye concentration observed on the shore (approximately 2 ppb, with a corresponding calculated on shore dilution value of approximately 36 to 1), is likely more representative of actual dilution conditions in that environment over the long term. Dye concentrations less than the 2 ppb (or dilution values greater than 36 to 1) were the result of the following:

- A lack of build up of dye in the near-field ambient waters resulting from the short duration dye releases;
- Time-dependant along-stream mixing of the dyed plume with undyed ambient water; and
- Far-field mixing in the surf zone that was highly variable.

All subsequent analysis, discussion and interpretation will therefore be based upon using the maximum measured dye concentration of 2 ppb (and the corresponding calculated minimum dilution value of 36 to 1).

5.6.5 COMPARISON OF THERMAL PLUME COMPUTER MODEL & DYE STUDY RESULTS

This section presents a comparison of the observed dye behavior with results from the computer model of the thermal plume. Additional computer modeling was conducted to assist in the understanding of plume dynamics and to update the predictions detailed previously on the basis of the dye field observations.

This section also includes a discussion on the apparent mechanics of the formation of the plume at its movement relative to the development of an "intermediate field" (not addressed by conventional computer modeling and how this impacts the prediction of dilution at downstream (or far-field) locations.

5.6.5.1 Limitations

Current data (speed and direction) was not obtained during the dye study. However, the uniformity / symmetry of the surfaced dye plume (as shown in **Figure A-2** indicates that ambient currents near the outfall during the dye study were minimal.

5.6.5.2 Data Synopsis

Dye Discharge

Rhodamine WT dye was introduced into the discharge vault to yield a measured "in-pipe" dye concentration of 72 ppb. The dyed effluent was released through the discharge pipe and photographed from the air with "time-series" photography.

The aerial photos of the dye discharges where enhanced to highlight the dyed effluent and to allow a determination of the variations in concentrations during the dye releases. Dye concentrations were assessed through analysis of the optical color scales of calibrated time-series photos of the dye field. The color code was calibrated with in-situ measurements from a fluorometer being towed through the dye field at the water surface.

Discharge Conditions

The discharge conditions from the AES HBGS at the time of the dye study were as follows:

- Number of pumps running 5 pumps;
- Discharge per pump 44,000 gpm;
- Total Discharge 220,000 gpm (13.88 m^3/s) (steady-state flow);
- Minimum effluent temperature 26.7 ^oC
- Maximum effluent temperature 29.6 ^oC
- Mean effluent temperature 28.2 ^oC

Observed Dilution During Discharge

The color variations from the photographic data and towed fluorometer data were correlated. The following dye concentrations were measured using the fluorometer based upon that correlation:

- The highest dye concentration was measured directly over the outfall terminus;
- The measured dye concentration above the terminus was 12 ppb, indicating the minimum initial dilution at that time was approximately 6 to 1;
- As the plume spread, the photos indicated that the dye concentrations at the edge of the circular plume approximately 600 feet (180 m) away from the center were in the order of 4 ppb indicating an additional dilution of 3 to 1;

Observations From Dye Study Photos

Observation of the time-series dye imagery between 15:41 and 15:56 are as follows:

- The plume surfaces;
- The plume is relatively symmetrical around the outfall;
- A slightly oblong pattern of reduced concentration at the plume perimeter occurs on the nearshore (shallower) side of the discharge;
- The highest concentration appeared to stay relatively centered and symmetrical over the outfall;
- The highest concentration appeared to grow proportionately with relatively symmetrical dissipation on the outer edges of the plume; and
- Edges of the readily identifiable plume (as shown previously in **Figure A-2**) had radiated as much as 620 feet (189m) in 14 minutes (approximately 0.18 to 0.23 m/s average velocity in all directions).

Dye Study Conclusions

The following conclusions can be drawn from the results of the dye study:

- The ambient currents at the time of the dye study were insufficient to significantly move the discharge from the immediate area of the outfall;
- A major component of the plume growth was due to physical displacement (discharge displacing the ocean water); and
- Dissipation of the plume was primarily driven through salinity and thermal gradients at the edge of the plume.

5.6.5.3 Comparison Of Dye Study Results & Initial Modeling Results

The initial computer modeling assumed that the plume moved laterally from the region of discharge towards a specific target area – in this case the shoreline – to predict possible concentrations at that target area.

The photographs of the dye study clearly indicate that the discharge "accumulates" locally, likely in three dimensions, and grows radially as the discharge mixes with the ambient water. Conventional modeling tools do not address this formation of an "intermediate" field (between near-field and far-field). Because of the disparity between the assumptions governing the model function, and the actual conditions, the UM model is not suitable to describe the "intermediate field" conditions that were observed during the dye study.

The UM model was designed to terminate calculation of the near-field conditions when the plume reaches the surface. The UM model inherently assumes that the far-field algorithms take over at the termination of the near-field calculations. Accordingly, the computer modeling previously conducted assumed that lateral movement of the thermal plume (and hence far-field analysis) commenced when the thermal plume surfaced, with relatively low initial dilution and a small diameter (field width for secondary dilution analysis).

Results from the dye study indicated that the discharge continues to displace the near-field area after surfacing, contributing to a substantially larger effluent field (intermediate-field). Presumably under the right oceanographic conditions, this larger field could be carried towards a target area. Under such circumstances, this consideration begs the question, "would the concentrations at impact be similar to those predicted by UM and Brooks?"

The photographs of the dye study provide valuable insight to the near field / intermediate-field mechanics of the plume "growth" to be used in determining initial conditions to answer the above question.

The results of the dye study were used as inputs to a far-field dilution model to develop a comparison between what the far-field concentrations would be with a wider, less concentrated plume and the narrow, highly concentrated plume.

5.6.5.4 Comparison Of Dye Study Results & Additional Computer Modeling

Supplementary Model Execution

The UM model generates an output containing:

- A tabulation of the input data;
- Numerically calculated parameters specific to each case; and
- The plume characteristics when it encounters the ocean surface.

The components of the output are user-definable. This feature was used for a detailed investigation of the dilution predictions relative to the mechanics of the plume observed in the dye study.

The most relevant output as it relates to this section is the predicted average dilution, which is less than the centerline dilution. The computer model, using the default calculation parameters, yielded a dilution prediction of 1.3 immediately above the discharge port, which would yield a concentration of 55 ppb based on an initial concentration of 72 ppb.

5.6.5.5 Comparison Of Dye Study Results & UM Model Results Relative To Plume Formation

Near Field Observations – Dye Study

On the basis of the observations made from the dye study photographs, the near field mechanics in the development of the effluent field appear to be as follows:

- Effluent discharge consists of heated saline water;
- The heated discharge is less dense than ambient ocean water at the point of discharge;
- Effluent is discharged vertically upwards, in the middle of the water column;
- The port diameter is roughly twice as large as the water depth over the port;
- The heated plume maintains its vertical momentum all the way to the surface;
- The plume spreads horizontally once it intersects the surface, flowing radially away from the center;
- The upward plume motion, at mid-depth in the water column, likely produces an upwelling and entrainment of the near-bottom waters beneath the vertical jet;
- The localized upwelling is replaced by a centripetal, near-bottom (or at least sub-plume) current pattern;
- At least a portion of the peripheral areas of the plume are likely "re-entrained" in the centripetal underflow, ultimately becoming part of the upwelling near the vertical discharge;
- The upwelling waters and the plume will intersect the surface together;
- The concentration of this "intermediate-field" is somewhat attenuated throughout the plume; and
- The plume is ultimately laterally displaced by the relatively weaker ambient currents.

As shown on **Figure A-2,** the initial field width of the plume has grown to an ellipse with a major axis diameter of approximately 360 feet (109 m), a minor axis (shoreward) diameter of approximately 230 feet (70 m) with a maximum measured dye concentration of 12 ppb.

Near Field Analysis – UM Model

The UM model stops the "near-field" calculation when the plume intersects the ocean surface. The UM model predicted a plume width of 21 feet (6.6 m) with an average concentration of 55 ppb (based on an initial dye concentration of 72 ppb).

At this time, we cannot account for the disparity between the UM predictions and the measured concentration directly over the discharge port. It is obvious that there are other influences (such as the upwelling or intrusion from centripetal currents induced by the recirculation of the plume) that are not factored into the algorithms of UMs near field calculation methods.

The UM model does not have the capability to generate the parameters of the "intermediate" or transitionary field that was observed during the dye study.

Far-Field Observations – Dye Study

Dye concentrations 600 feet from the plume center were measured from 1 ppb to 4 ppb indicating a secondary dilution (dilution due to dispersion) of approximately 2 (relative to an average concentration of 8 ppb at the cessation of initial dilution).

The ambient current (which dictates the time of travel to a downstream location) was not confirmed.

Images 177 and 181 through 183, as shown previously on **Plate 4-19a,** also show the dye remnants from Dye Release #3 at a substantial distance away from the surfacing plume as part of Dye Release #4. It is apparent that the plume from Dye Release #3 had radiated a significant distance from the source, indicating that a higher far-field dispersion factor should be used.

Far-Field Analysis – UM Model

The radial rate of expansion of the plume will, at some point in time, reach equilibrium with the ambient currents. At that point, the plume will start to be influenced by the relatively slow ambient currents which then triggers the initiation of the two dimensional analysis of secondary dilution. Where this "equilibrium" occurs is difficult to predict.

However, on the basis of conservation of mass, momentum and energy (the UM model), and the geometric expansion of the plume predicted and governed by the Brooks algorithm, it is logical to expect that the downstream predictions should be realistic regardless of the formation of the intermediate-field.

It is also logical to expect that the "far-field" concentrations, based on the small concentrated plume predicted by UM, will be comparable to the "far-field" concentrations of a wider, less-concentrated plume (as observed during the dye study).

The plume configuration as shown on **Figure A-2,** was assumed to be the initial condition for this subsequent secondary dilution analysis. The algorithms used in UM (Brooks algorithm, using the 4/3 power law) and a far-field dispersion factor of 0.000453 $m^{2/3}/s$ was applied to an initial field width of 360 feet (110 m). This width corresponds to the long axis of the plume shown in **Figure A-2,** with a measured initial average dye concentration of 8 ppb over that plume.

The Brooks model cannot be run separately from UM. Accordingly, a separate program was developed expressing:

$$C_{max} = c_o \, erf\{[3/2/[(1+2/3\beta x/B)^3-1]]^{1/2}\}$$

Where: C_{max} = Maximum concentration at the centerline of the dye field;
c_o = Average concentration of the plume at the start of dispersion;
$\beta = 12\alpha B^{4/3}/uB$;
B = Initial field width (450 feet or 137 m from **Figure A-2);**
X = Downstream distance (varies from 600 to 1800 feet for comparative purposes); and
$\alpha = 0.000453 \; m^{2/3}/s$.
(In the absence of site measurements for α, this is the highest number that can be justified)

The analysis was completed using data from the dye field imagery to generate downstream concentrations at various distances. The analysis was repeated using the near-field parameters generated in UM. The comparative input data are summarized as follows in **Table 5-7**:

Table 5-7 - Comparison Of Model Input Data

Parameter	Dye Study Observations	UM Data
Initial average concentration	8 ppb	55 ppb
Initial field width	361 feet (110 m)	22 feet (6.6m)
Ambient current	0.05 m/s to 0.15 m/s	0.05 m/s to 0.15 m/s
Downstream distances	600 to 1800 feet (180 to 540 m)	600 to 1800 feet (180 to 540 m)

Predicted concentrations at given downstream distances are shown graphically on **Figure A-3.**

Analysis Of Results

Figure A-3 shows the concentrations at various downstream locations for the two conditions analyzed (narrow "near-field" width with high concentrations predicted by UM, and the wide intermediate plume with low concentrations actually observed during the dye study). The concentrations are shown to be comparable at downstream locations, leading to the conclusion that in spite of the unusual mechanics of the development of the wide transitionary field, downstream predictions using the UM / Brooks combination model are relatively comparable:

- At low currents, the downstream concentrations at a plume centerline are comparable when using the actual concentrations and widths observed in the dye field and the UM initial dilution calculations; and
- As current speeds increase, the distance downstream to comparable centerline concentrations increase (note that the "crossover" point in the graphs occurs further downstream with faster currents).

The predicted concentrations were computed on the basis of time (downstream distance divided by current velocity) for each case and summarized graphically on **Figure A-4.** The top portion of the figure presents the actual predictions while the bottom portion of the figure summarizes the differences in the predictions as a function of time. The calculations indicate that after one hour of travel, the difference between the far-field concentration predictions and the measured dye concentrations vary by less than 5% of each other. A more conservative (smaller) dispersion factor increases the time to comparable concentrations, while a less conservative (larger) dispersion factor shortens the time.

Relevance Of The UM / Brooks Model

Based on the comparison of the graphical representations of the predicted **concentrations** and **differences in concentration** on **Figure A-3** and **Figure A-4** respectively, it is concluded that the UM / Brooks model, yields a downstream prediction that is relatively comparable to that observed during the dye study **when the plume is moved laterally by an ambient current**.

The additional computer modeling that was performed as part of this section used a far-field dispersion coefficient that was less conservative (larger) than the one used in the original computer modeling analysis, but use of this larger coefficient appears to be justified on the basis of the apparent plume spread shown in **Figure A-2.**

Dye Study & Computer Modeling Comparison Conclusions

- The UM program terminates calculation when the plume hits the water surface, but the plume is still in "jet" configuration when it hits the surface and has not fully developed. The UM model is not well suited to predicting initial dilution in a comparatively shallow (water depth less than ½ the port diameter) mid-water discharge depth when considering a highly buoyant, large diameter plume;
- During relatively stagnant conditions (low velocity ambient currents), the plume grows radially with dispersion occurring at the outer edge of the circular field. Under these conditions, the plume is unlikely to be flushed from the area at a rate greater than the supply of the effluent into the zone of initial dilution and thus the plume forms a transitionary or "intermediate" field that cannot adequately be characterized by the UM model;
- As a result of the two points noted above, the UM model predicts substantially lower initial dilution values than those that were actually measured on the ocean surface during the dye study; and
- When "far-field" analysis using the Brooks equation with a far-field dispersion coefficient typical of open ocean conditions is applied to the intermediate field, actual downstream dye concentrations measured during the dye study correlate well to those predicted with the UM / Brooks model.

5.7 BACTERIA AND THE RECEIVING ENVIRONMENT

5.7.1 AB 411 SINGLE SAMPLE CRITERIA

5.7.1.1 Bacterial Concentrations Required at DV to Cause AB 411 Single Sample Exceedance at 9N

The minimum dilution value measured at various monitoring stations along the beach was approximately 36 to 1. On the basis of examining a dilution value that was calculated from a dye concentration measurement of 2 ppb, the concentrations of indicator bacteria that would be required in the AES HBGS discharge vault to cause an AB 411 single-sample exceedance on the beach have been calculated.

Figure 5-8 represents the total coliform concentration required in the discharge vault to cause an exceedance on the beach. For example, the AB 411 single-sample criteria for total coliform is 10,000 cfu / 100 ml. In order to measure that concentration on the beach (assuming that no other sources are contributing), and based on the dilution value calculated from the outfall to the

beach of 36 to 1, the required concentration of total coliform in the discharge vault would be approximately 360,000 cfu/100 ml. **Figures 5-9** and **Figure 5-10,** respectively, represent the required concentrations of fecal coliform and Enterococcus to cause an exceedance on the beach based upon their AB 411 single-sample criteria limits. A tabular summary of the key concentrations required in the discharge vault is shown below in **Table 5-8**

Table 5-8 - Concentrations Of Indicator Bacteria Required In The DV To Cause An AB 411 Single Sample Exceedance On The Beach

Category	Minimum Dilution (36 to 1)
Total Coliform	360,000
Fecal Coliform	14,400
Enterococcus	3,744

All concentrations are expressed in units of cfu/100ml.

5.7.1.2 Potential Contribution of DV Indicator Bacteria to Beach Water Quality Degradation

The water in the discharge vault was sampled extensively from July 16, 2002 through October 15, 2002, for indicator bacteria including total coliform, fecal coliform and Enterococcus. Water in the discharge vault is discharged through the outfall, diluting and mixing with the ocean. Portions of this mixture are then transported back onto the beach by outfall-driven and wind-driven currents through the surf zone. However, if a water sample was collected from a shoreline monitoring station on a particular day and that sample was found to contain 10,000 cfu/100 ml of total coliform (the AB 411 single-sample criteria), then what fraction of bacteria in that water sample could possibly be attributed to the discharge from the AES HBGS? Or how much impact does the DV discharge have on beach water quality along Huntington State Beach?

Temporal indicator bacteria data for DV coupled with dilution ranges measured at various locations along the beach during the offshore dye study have been used to assess the potential percentage contribution of the AES HBGS to AB 411 single sample exceedances at 9N. **Figure 5-11** shows temporal plot of total coliform bacteria concentrations, the concentrations represent the percentage contribution from the AES HBGS discharge towards an AB 411 single-sample exceedance on the beach. The predictions are based upon measured concentrations of bacteria

in the discharge vault and the predicted minimum dilution of 36 to 1 for the outfall plume. For example, on August 27, 2002, a total coliform concentration of 1,700 cfu/100 ml was measured in the discharge vault. Using the minimum dilution value of 36 to 1, the expected total coliform concentration on the beach as a result of that sample is approximately 47 cfu/100 ml. 47 out of 10,000 represents a percentage contribution of approximately 0.5% as shown on the figure. From **Figure 5-11**, it can be seen that the maximum potential contribution of total coliform from the DV during the study period towards an AB 411 single sample exceedance was approximately 0.5%.

Figure 5-12 and **Figure 5-13**, show percentage contributions towards AB 411 single sample exceedances for fecal coliform and Enterococcus, respectively. **Figure 5-12** shows that the maximum potential contribution of fecal coliform from the DV during the study period towards an AB 411 single sample exceedance was approximately 3%. **Figure 5-13** shows that the maximum potential contribution of Enterococcus from the DV during the study period towards an AB 411 single sample exceedance was approximately 16.3%. These maximum values were based upon the lowest calculated dilution values from the outfall to the beach.

5.7.1.3 Net Flux of Indicator Bacteria From the AES HBGS Into the Receiving Environment

Water samples were also collected from IV as well as DV and analyzed for indicator bacteria including total coliform, fecal coliform and Enterococcus. These indicator bacteria were found to be present in water being drawn into the AES HBGS. Since the water in the IV had indicator bacteria present, as did the water being discharged from the DV, a net bacterial flux has been derived to determine if the AES HBGS is increasing or decreasing the net bacterial load to the receiving waters of Huntington State Beach. Indicator bacteria concentrations measured in the IV were subtracted from those measured in the DV at DV-10, the resultant difference represents the net bacterial flux of the AES HBGS. Samples from the IV and DV10 were generally taken within one hour of each other (the average lag time between samples for the study period is 25 minutes). The minimum dilution value of 36 to 1, detailed previously, was again used to calculate the percentage contribution of that net flux towards an AB 411 single-sample exceedance on the beach.

Figure 5-14 shows the net flux of total coliform as a function of time and percentage contribution towards an AB 411 single-sample exceedance. From the figure it can be seen that the AES HBGS does increase the net load of total coliform bacteria to the receiving waters of Huntington State Beach, but relative to the AB 411 single-sample criteria, that load is very small

(a maximum contribution of approximately 0.4% towards a single-sample exceedance on the beach).

Figure 5-15 shows the net flux of fecal coliform bacteria from the AES HBGS. Again, it can be seen that the AES HBGS does increase the net load of fecal coliform bacteria to the receiving waters, but the maximum contribution was approximately 3.0% towards an AB 411 single-sample exceedance for the minimum dilution values observed during the study period.

Figure 5-16 shows the net flux of Enterococcus bacteria from the AES HBGS. This figure is particularly interesting because while it does show periodic spikes of net increases of Enterococcus to the receiving environment, it shows more spikes as net decreases to the receiving waters. Basically, for the study period, and on the basis of measured concentrations of Enterococcus in the IV and DV10, the IV water contained more Enterococcus than the DV10 water. Enterococcus was being removed from the environment during those times. For the Enterococcus spikes indicating a net increase in the receiving environment, the maximum percentage contribution to an AB 411 single-sample exceedance (based on the minimum observed dilution values) was approximately 11%.

5.7.2 AB 411 30-DAY GEOMETRIC MEAN CRITERIA

5.7.2.1 Bacterial Concentrations at DV to Cause AB 411 30-Day Geometric Mean Exceedance at 9N

The concentrations of indicator bacteria that would be required in the AES HBGS discharge vault to cause an AB 411 30-day geometric mean exceedance on the beach have been calculated.

Figure 5-17 represents the total coliform concentrations required in the discharge vault to cause an exceedance on the beach. For example, the AB 411 30-day geometric mean criteria for total coliform is 1,000 cfu/100 ml. In order to measure that concentration on the beach (assuming that no other sources are contributing), and based on the dilution value calculated from the outfall to the beach of 36 to 1, the required concentration of total coliform in the discharge vault would be approximately 36,000 cfu/100ml. **Figures 5-18** and **Figure 5-19,** respectively, represent the required concentrations of fecal coliform and Enterococcus to cause an exceedance on the beach based upon their AB 411 30-day geometric mean criteria limits. A tabular summary of the key concentrations required in the discharge vault is shown below in **Table 5-9**

Table 5-9 - Concentrations Of Indicator Bacteria Required In The DV To Cause An AB 411 30-Day Geometric Mean Exceedance On The Beach

Category	Minimum Dilution (36 to 1)
Total Coliform	36,000
Fecal Coliform	7,200
Enterococcus	1,260

All concentrations are expressed in units of cfu/100 ml.

5.7.2.2 Potential Contribution of DV Indicator Bacteria to Beach Water Quality Degradation

Figure 5-20 shows temporal plot of total coliform bacteria concentrations, the concentrations represent the percentage contribution from the AES HBGS discharge towards an AB 411 30-day geometric mean exceedance on the beach. The predictions are based upon measured concentrations of bacteria in the discharge vault and the predicted minimum dilution of 36 to 1 for the outfall plume. For example, on August 27, 2002, a total coliform concentration of 1,700 cfu/100ml was measured in the discharge vault. Using the minimum dilution value of 36 to 1, the expected total coliform concentration on the beach as a result of that sample is approximately 47 cfu/100 ml. 47 out of 1,000 represents a percentage contribution of approximately 4.7% as shown on the figure. From **Figure 5-20**, it can be seen that the maximum potential contribution of total coliform from the DV during the study period towards an AB 411 30-day geometric mean exceedance was approximately 4.7%.

Figure 5-21 and **Figure 5-22**, show percentage contributions towards AB 411 30-day geometric mean exceedances for fecal coliform and Enterococcus, respectively. **Figure 5-21** shows that the maximum potential contribution of fecal coliform from the DV during the study period towards an AB 411 30-day geometric mean exceedance was approximately 6%. **Figure 5-13** shows that the maximum potential contribution of Enterococcus from the DV during the study period towards an AB 411 exceedance was approximately 49%. These maximum values were based upon the lowest calculated dilution values from the outfall to the beach.

5.7.2.3 Net Flux of Indicator Bacteria From the AES HBGS Into the Receiving Environment

The minimum dilution value of 36 to 1, detailed previously, was again used to calculate the percentage contribution of the net flux towards an AB 411 30-day geometric mean exceedance on the beach.

Figure 5-23 shows the net flux of total coliform as a function of time and percentage contribution towards an AB 411 30-day geometric mean exceedance. From the figure it can be seen that the AES HBGS does increase the net load of total coliform bacteria to the receiving waters of Huntington State Beach with a maximum contribution of approximately 4% of the 30-day geometric mean criteria.

Figure 5-24 shows the net flux of fecal coliform bacteria from the AES HBGS. Again, it can be seen that the AES HBGS does increase the net load of fecal coliform bacteria to the receiving waters with a maximum contribution of approximately 6% towards an AB 411 30-day geometric mean exceedance for the minimum dilution values observed during the study period.

Figure 5-25 shows the net flux of Enterococcus bacteria from the AES HBGS. Again, as with the single-sample criteria figure (**Figure 5-16**), while this figure does show periodic spikes of net increases of Enterococcus to the receiving environment, it shows more spikes as net decreases to the receiving waters. Enterococcus was being removed from the environment during those times. For the Enterococcus spikes indicating a net increase in the receiving environment, the maximum percentage contribution to an AB 411 30-day geometric mean exceedance (based on the minimum observed dilution values) was approximately 34%.

5.7.3 MICROBIAL SOURCE TRACKING

Microbial source tracking was conducted because standard microbiological indicators may originate from a variety of warm-blooded animals, and there is frequently a concern that high bacterial indicator values may not necessarily indicate human fecal contamination.

5.7.3.1 Bacteroides/Prevotella Analysis

Microbial source tracking techniques employed in this study were based upon a genetic test for the DNA from a group of strictly anaerobic bacteria (in the genera *Bacteroides* and *Prevotella*) that are very common in human feces but not demonstrated to originate or grow elsewhere (Bernhard and Field, 2000). In this study the method was used to show whether or not human fecal contamination can be found in a particular sample, hence the result should be used in

concert with other indicators. The bacterial data indicates that the AES HBGS was not a source of human-derived contamination during this study, but it probably entrained contaminated water from the ocean through its cooling system **(Table 4-10)**.

5.7.3.2 Enterovirus Analysis

Human pathogenic viruses are microbiological indicators from human rather than animal origin. Viruses are different from bacteria in their composition and also basic biological properties. The enteroviruses examined in this study represent a "family" of related viruses (including poliovirus, echoviruses, and coxsackieviruses) that can all be detected simultaneously because they share genetic elements. They are believed to derive from human fecal contamination only.

The interpretation is similar to that from the bacterial tests – the AES HBGS did not seem to be a significant source of human-derived contamination during this study period. Enteroviruses were not found in the samples that tested positive for human *Bacteroides/Prevotella*, this is not necessarily a contradiction, as it is common to have poor correlations between bacterial and viral contamination (e.g. Noble and Fuhrman 2001). The causes may be due to differences in relative amounts of these microbes in source material, differential transport and survival of bacterial and viral contaminants, and/or possibly different detection thresholds in the assays.

5.7.3.3 What Happened in 2001?

The absence of high bacterial counts within the AES HBGS in the summer of 2002, contrasts with the data observed during the previous summer, when high bacterial counts were recorded within the discharge vault (and other locations). Ocean water quality has not differed significantly between the two years. Using comparable data from January 1, 2001; and January 1, 2002; to July 29, 2000, 2001 and 2002; water quality (as indicated by number of exceedances of AB411) has not altered significantly. In the limited 2000 period 11 AB 411 exceedances were recorded; nine and 13 for the corresponding periods in 2001 and 2002 respectively. The time periods were chosen to reflect the start-up of chlorination by the OCSD in August 2002. Consequently internal practices within the AES HBGS (active retooling and construction) were most likely responsible for the high values observed in 2001 within the AES HBGS. It appears that the internal practices were not repeated in the summer of 2002. Furthermore, during 2001 the number of operational pumps was considerably lower and occasionally at zero by comparison with 2002. During these periods little bacterial flushing would occur and concentrations could increase as loading from freshwater sources continued.

5.7.3.4 What is Happening Now?

The OCSD has completed the installation of the discharge chlorination and dechlorination and is now attaining an effective high kill rate on the bacteria present in the discharge waster stream. Bacterial exceedances along Huntington State Beach are still periodically occurring, although not to the same magnitude as the 1999 event. Other likely sources are currently being investigated, including the Santa Ana River and other potential sources of urban run-off. During the intensive two-week sampling period in August, Komex teamed with Dr. Jones and Fuhrman of USC, Sierra Analytical Laboratories of Laguna Hills, the City of Huntington Beach, and the City of Huntington Beach Police Department and their Aero Bureau to conduct a non-funded investigation into another alternate source in the region of Huntington State Beach.

Thermal-Plume Onshore Transport (T-POT) Hypothesis

Four hypotheses have been investigated and evaluated as part of the scope of this project. Three of the four hypotheses have been rejected and the fourth, whilst accepted, has not been considered as a significant contributor to surf zone water quality degradation centered at water quality monitoring station 9N (Station 9N) on Huntington State Beach. However, the thermal plume produced by the AES HBGS discharge outfall may have a role in transporting pre-contaminated water to shore and hence could be a key component of observed surf zone water quality impairment at Station 9N (TAG, 2003). Urban runoff from several water bodies in the vicinity of the study area has been shown to travel through the surf-zone and then move up- or down-coast, relative to prevailing coastal currents (Jones and Svejkovsky, 2000). Depending on the urban runoff source location and the prevailing currents just outside of the surf-zone, urban runoff can be transported into the vicinity of Station 9N and the AES HBGS discharge. The T-POT mechanism observed in this study and suggested by the May 2000 study (Jones and Svejovsky, 2000), results in a convergence between the radial flow of the cooling water discharge and the alongshore coastal current. The convergence results in the shoreward transport on the upcurrent side of the cooling water through the surfzone and into the beach area around Station 9N (Illustrated below). The T-POT mechanism was observed during the dye studies conducted on the thermal discharge during this study, however it should be noted that the dye studies were only performed during down-coast flow, and supporting observations for an analogous effect for the up-coast flow (as illustrated) were not available.

Thermal-Plume Onshore Transport Mechanism (T-POT)

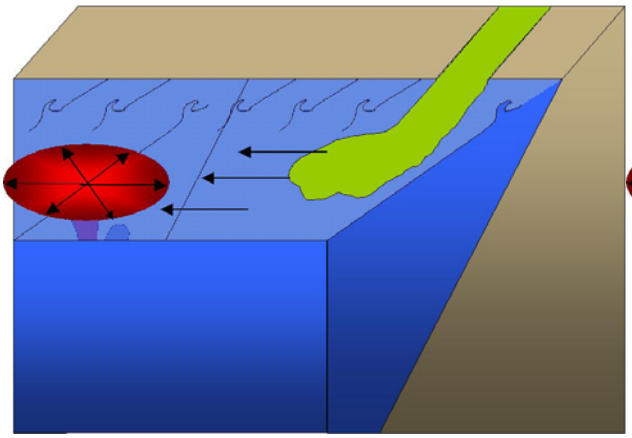


Illustration 1: During ebb tides, urban run-off from the Santa Ana River (green) flows out through the surfzone and is advected upcoast towards the AES HBGS discharge plume (red) by upcoast currents.

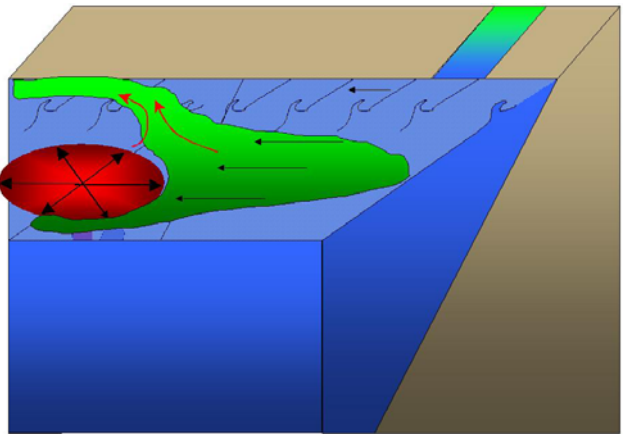


Illustration 2: Once in the vicinity of the AES HBGS thermal discharge, the plume of urban run-off is focused as it converges with the radial thermal plume. The portion of the plume between the discharge and the beach is transported through the surfzone and onto the beach.

The T-POT mechanism would be consistent with the higher bacteria concentrations observed during spring tides when flushing from the runoff sources is larger (Brocard, 2003). The T-POT mechanism would also explain localized chronic surfzone water quality impairment at 9N and the apparent disconnect with the source of the bacteria (Relatively good surf zone water quality at other monitoring stations between 9N and the source of the bacteria). An investigation into the T-POT mechanism was beyond the scope of this investigation, but it is recommended that additional work be performed to attempt to qualify or quantify the significance of the T-POT mechanism on beach water quality during up-coast and down-coast flows (**Recommendations Section 7**).

The table below shows bacterial concentration data for all water quality samples collected from the AES HBGS from September 3 through September 7, 2002.

The data has been sorted chronologically by sample location (BCR, BFW, BSW, SWS, GP, IV, DV-0 and DV-10) with columns including the sample ID, sample location, sample date, enterococci concentration, fecal coliform (FC) concentration, total coliform (TC) concentration, and FC/TC ratio. All samples that exceed the single-sample criteria have been color-coded (bold text and yellow shading for enterococci greater than 104 CFU/100 mL, bold text and green shading for fecal coliform greater than 400 CFU/100 mL, and bold text and dark blue shading for total coliform greater than 10,000 CFU/100 mL).

Sample ID	Sample Location	Date	ENT	Fecal Coliform	Total Coliform	FC/TC Ratio
BCR-0903	Blackford's Ditch (BD)	09/03/02	**330**	210	8,000	0.03
BCR-0904	Blackford's Ditch (BD)	09/04/02	**300**	60	7,000	0.01
BCR-0905	Blackford's Ditch (BD)	09/05/02	**280**	**600**	**11,000**	0.05
BCR-0906	Blackford's Ditch (BD)	09/06/02	**110**	50	4,000	0.01
BCR-0907	Blackford's Ditch (BD)	09/07/02	**500**	**600**	**12,000**	0.05
BFW-0903	Boiler Fireside Wash (BFW)	09/03/02	**3,000**	**580**	900	**0.64**
BFW-0904	Boiler Fireside Wash (BFW)	09/04/02	**1,800**	**2,200**	**5,400**	**0.41**
BFW-0905	Boiler Fireside Wash (BFW)	09/05/02	**1,700**	**500**	**5,000**	**0.10**
BFW-0906	Boiler Fireside Wash (BFW)	09/06/02	**17,000**	**900**	**17,000**	0.05
BFW-0907	Boiler Fireside Wash (BFW)	09/07/02	**35,000**	**900**	**67,000**	0.01
BSW-0903	Boiler Sump Wash (BSW)	09/03/02	**1,200**	**520**	**1,900**	**0.27**
BSW-0904	Boiler Sump Wash (BSW)	09/04/02	80	130	800	**0.16**
BSW-0905	Boiler Sump Wash (BSW)	09/05/02	**130**	110	2,000	0.06
BSW-0906	Boiler Sump Wash (BSW)	09/06/02	60	50	220	**0.23**
BSW-0907	Boiler Sump Wash (BSW)	09/07/02	**140**	86	300	**0.29**
SWS-0903	Storm Water Sump (SWS)	09/03/02	70	30	90	**0.33**
SWS-0904	Storm Water Sump (SWS)	09/04/02	**130**	**570**	600	**0.95**
SWS-0905	Storm Water Sump (SWS)	09/05/02	20	**600**	**1,100**	**0.55**
SWS-0906	Storm Water Sump (SWS)	09/06/02	**930**	**820**	**2,800**	**0.29**
SWS-0907	Storm Water Sump (SWS)	09/07/02	80	340	700	**0.49**
GP-0903	General Retention Basin (GP)	09/03/02	40	30	5,300	0.01
GP-0904	General Retention Basin (GP)	09/04/02	60	80	9,000	0.01
GP-0905	General Retention Basin (GP)	09/05/02	30	50	5,000	0.01
GP-0906	General Retention Basin (GP)	09/06/02	50	120	5,000	0.02
GP-0907	General Retention Basin (GP)	09/07/02	30	50	**32,000**	0.00
IV-0903	Intake Vault (IV)	09/03/02	0	1	2	**0.50**
IV-0904	Intake Vault (IV)	09/04/02	5	9	27	**0.33**
IV-0905	Intake Vault (IV)	09/05/02	1	0	3	0.00
IV-0906	Intake Vault (IV)	09/06/02	1	3	8	**0.38**
IV-0907	Intake Vault (IV)	09/07/02	1	1	2	**0.50**
DV-0-0903	Discharge Vault (DV0)	09/03/02	0	10	250	0.04
DV-0-0904	Discharge Vault (DV0)	09/04/02	16	30	600	0.05
DV-0-0905	Discharge Vault (DV0)	09/05/02	2	10	300	0.03
DV-0-0906	Discharge Vault (DV0)	09/06/02	3	5	300	0.02
DV-0-0907	Discharge Vault (DV0)	09/07/02	3	8	200	0.04
DV-10-0903	Discharge Vault (DV10)	09/03/02	0	5	100	0.05
DV-10-0904	Discharge Vault (DV10)	09/04/02	0	0	0	--
DV-10-0905	Discharge Vault (DV10)	09/05/02	0	2	500	0.00
DV-10-0906	Discharge Vault (DV10)	09/06/02	2	2	80	0.03
DV-10-0907	Discharge Vault (DV10)	09/07/02	2	6	40	**0.15**

LEGEND

280	Enterococci concentration that exceeds AB 411 single-sample criteria of 104.
2,200	Fecal coliform concentration that exceeds AB 411 single-sample criteria of 400.
11,000	Total coliform (TC) concentration that exceeds AB 411 single-sample criteria of 10,000.
5,400	TC that exceeds AB 411 single-sample criteria of 1,000 when FC/TC ratio is > 0.1.
0.41	Fecal coliform to total coliform (FC/TC) ratio that exceeds 0.1.
ENT	Enterococcus

Bacterial units as colony forming units or equivalent per ml.

Additionally, the FC/TC ratios have been calculated for the data set. All FC/TC ratios that exceed 0.1 have been color-coded with bold text and light gray shading. For the total coliform samples that had an FC/TC ratio of 0.1 or greater, and had a total coliform concentration greater than 1,000, cells have been color-coded with bold text and light blue shading.

The data for this period shows that bacteria (all three types) were found in concentrations that routinely exceeded the AB 411 single-sample criteria in Blackford's Ditch (BCR), the boiler fireside wash (BFW), the boiler sump wash (BSW), and the storm water sump (SWS), with elevated concentrations of total coliform also being found in the general retention basin (GP). However, bacteria concentrations were low in the intake vault (IV) samples as well as the discharge vault samples (both DV-0 and DV-10).

For the time period shown, the data shows that the AES HBGS was neither taking in nor discharging water that was appreciably "contaminated" with bacteria. Water quality data for Station 9N for this same time period shows the following:

Station	Date	Enterococci (CFU/100 mL)	Fecal Coliform (CFU/100 mL)	Total Coliform (CFU/100 mL)
9N	03-Sep-02	34	20	40
9N	04-Sep-02	52	80	80
9N	05-Sep-02	38	20	20
9N	06-Sep-02	70	500	500
9N	07-Sep-02	70	500	500

At Station 9N it can be seen that measurable concentrations of all three types of bacteria were detected during this time period. In fact, the fecal coliform concentrations exceeded the AB 411 single-sample criteria of 400 on September 6 and September 7, 2002.

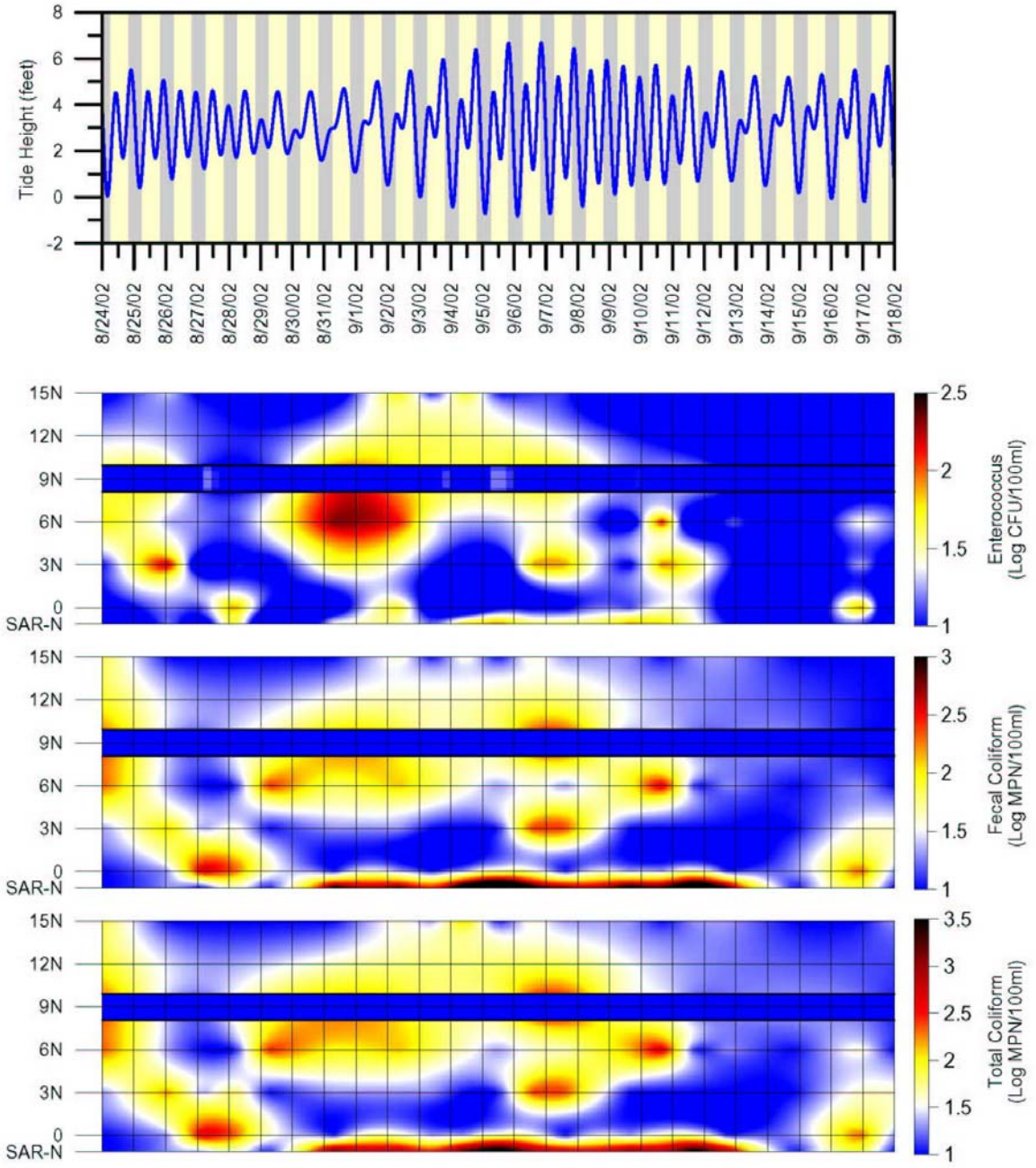
In considering results from the dye study and utilizing a minimum dilution of 36 to 1 from the outfall to the beach, the percentage contributions of the AES HBGS (for this time period) to the measured concentrations on the beach at Station 9N are in the table below.

AES HBGS discharge contribution at 9N during this period was negligible for enterococci and fecal coliform. The contribution of total coliform seems to be more significant strictly on the basis of the simple dilution ratio of 36 to 1 and the relative concentrations in the AES HBGS discharge and those measured at Station 9N, but they don't seem to be linked directly to the enterococci and fecal coliform proportions for the same two locations. The explanation for this

is unknown, but is probably related to the low absolute numbers of bacteria detected at these locations.

Elevated bacteria concentrations were detected at Station 9N during this period, the AES HBGS discharge contribution to those shoreline concentrations for enterococci and fecal coliform was negligible. The enterococci and fecal coliform bacteria measured at Station 9N during this time period are most likely from other sources and not the AES HGBS, as illustrated in the figure below. The figure is a graphical representation of bacterial concentrations along the coastline from the Santa Ana River to 15N for Enterococcus, Total coliform and Fecal coliform, between August 24 and September 18, 2004. Bacterial concentrations from the discharge vault have been overlaid at the 9N position (blue band) for comparison.

Temporal bacteria concentrations between Santa Ana River and 15N



6 CONCLUSIONS

6.1 HYPOTHESIS 1: SUB-THERMOCLINE ENTRAINMENT

Is sub-thermocline freshwater drawn into the AES HBGS intake, or entrained into the thermal discharge of the outfall pipe?

On several occasions cooler thermocline water was observed at the intake mooring (**Figure 5-4 and 5-5**). However, it would not appear to be a significant transport mechanism for thermocline or sub-thermocline water. Bacterial contamination of Huntington Beach has been a chronic problem and based on the low frequency and minimal extent that cooler thermocline or sub-thermocline water was observed at the intake mooring, entrainment of thermocline or sub-thermocline water by AES HBGS would not appear to be a significant mechanism (**Figure 4-33**). Bacterial indicator concentrations in the intake vault and at surfzone station 9N were occasionally higher than ambient concentrations on the four events that thermocline water was identified at the intake mooring (**Table 5-6**). Specifically, of the nine samples collected from the intake vault during suspected thermocline events; three of the total coliform samples were above average, two of the fecal coliform samples were above average and one of the Enterococcus samples was above average. The Enterococcus value was particularly elevated and represented an exceedance of AB 411 (**Table 5-6**). Surfzone water quality samples were also impaired compared to average; with two total coliform concentrations above 100 cfu/100ml, one fecal coliform concentration greater than the AB 411 standard of 400 cfu/100ml (500 cfu/100ml on September 7, 2002) and three fecal coliform concentrations greater than 50 cfu/100ml, and six of the nine Enterococcus values greater than 30 cfu/100ml (Note, neither total coliform or Enterococcus concentrations exceeded AB 411).

6.2 HYPOTHESIS 2: PLANT BACTERIAL PROLIFERATION

Are bacteria introduced into the AES HBGS intake from the ocean (at low concentrations), and selectively cultured by the elevated temperature of the cooling water in the plant?

Hypothesis 2 has been rejected because bacterial concentrations have not been demonstrated to increase significantly between the intake vault and the discharge vault (**Figure 4-12, Figure 4-13 and Figure 4-14**). Intake vault and discharge vault comparisons are dependant on which one of the three sample locations in the discharge vault are selected. Samples collected at water surface level in the discharge vault were either equivalent or greater in indicator bacteria

concentrations than samples collected in the intake vault, a similar trend was observed with the samples collected from DV-10. The data set for the DV-30 sample location was smaller, but the bacterial concentrations at this depth were consistently lower than both the intake vault and the other two sample locations for all three microbial analytes.

The net flux of total coliform, fecal coliform and Enterococcus through the AES HBGS was determined (**Section 5.7 and Section 6.2.1, 6.2.2, and 6.2.3 below**).

6.2.1 TOTAL COLIFORM

The AES HBGS did increase the net load of total coliform bacteria to the receiving waters of Huntington State and City Beaches (**Figure 5-14)**. Relative to the AB 411 single-sample criteria, that load is very small (a maximum contribution of approximately 0.4% towards a single-sample exceedance on the beach).

6.2.2 FECAL COLIFORM

The AES HBGS did increase the net load of fecal coliform bacteria to the receiving waters of Huntington State and City (**Figure 5-15).** The maximum contribution was approximately 3.0% towards an AB 411 single-sample exceedance for the minimum dilution values observed during the study period.

6.2.3 ENTEROCOCCUS

The AES HBGS did not increase the net load of Enterococcus bacteria to the receiving waters of Huntington State and City Beaches **(Figure 5-16)**. Unlike total and fecal coliform bacteria however, Enterococcus fluxes were more variable, with both net increases and decreases of Enterococcus to the receiving environment, with more spikes as net decreases to the receiving waters (**Figure 5-16**). Generally the IV contained more Enterococcus than DV10, and Enterococcus appeared to be removed from the environment with passage through the AES HBGS. For the Enterococcus spikes indicating a net increase in the receiving environment, the maximum percentage contribution to an AB 411 single-sample exceedance (based on the minimum observed dilution values) was approximately 16%.

The microbial contribution of surface freshwater flows was included in the net flux assessment and therefore these numbers do not represent purely microbial growth of entrained oceanic bacteria. It should be noted that a limited number of heat treatments within the AES HBGS co-incided with water quality sampling events.

6.3 HYPOTHESIS 3: LAND-BASED SOURCES

Are land-based sources of bacteria entering the discharge vault in the AES HBGS and are discharged to the ocean causing bacterial contamination of the surf zone of Huntington State Beach.

Hypothesis 3 has been accepted, whilst land-based sources of bacteria are entering the discharge vault via Blackford's Ditch and associated storm drains (**Table 4-1, Table 4-2 and Table 4-5; and Figure 4-5a and Figure 4-5b**) and the GP (**Table 4-1 and Table 4-2; and Figure 4-9**), the concentrations are not sufficiently high enough to cause impairment of the surf zone water quality at Huntington State or City of Huntington Beaches. During dry weather flow the **maximum** contribution of bacteria from the AES HBGS to bacterial counts at 9N has been calculated at 16% of the total number. During the three-month study period, potential AES HBGS bacterial contribution was approximately 0.5% of the total bacteria concentrations observed at 9N (**Figure 5-14, Figure 5-15 and Figure 5-16**).

6.4 HYPOTHESIS 4: SANITARY SEWERS

Are sanitary sewers within the AES HBGS connected to, or leaking into, the discharge vault?

Hypothesis 4 has been rejected; during the course of the sanitary dye study no dye was observed in the discharge vault (**Figure 4-23 and Section 4.4**). Furthermore throughout the daily and intensive sampling periods high concentrations of the three bacterial indicators were not observed in the discharge vault at all three sample locations (**Tables 4-1 and 4-2**). If either a continuous or periodic discharge from the sanitary sewer had occurred during the three months of sampling, elevated concentrations of all three-indicator bacteria would have been observed. Consequently the sanitary sewers are not connected to the discharge vault or any underground conveyances that discharge into the discharge vault.

7 RECOMMENDATIONS

Further investigation of the role of the AES HBGS facility as a source of bacteria in surf zone water quality of Huntington State Beach is not recommended at this time because of the microbial and oceanographic results presented in this study. Further investigation into the relationship between the AES HBGS discharge and other sources of bacteria within the coastal environment of Huntington State Beach is warranted.

As a general principle, discharges of fresh-water wastestreams through the AES HBGS discharge vault and outfall should be minimized.

The following remedial actions should be taken to improve the surf zone water quality of Huntington State Beach:

- Blackford's Ditch, the PCH/Newland Storm drain and the drain in the wildlife sanctuary car park should be disconnected from the AES HBGS discharge vault;
- The discharge vault should be appropriately sealed to prevent any backflow of water into the storm drains;
- Review of historic water quality data and in-plant heat treatments subsequent to this investigation to confirm that heat treatments do not provide a source of bacteria;
- Non-sanitary waste generated on site should undergo treatment to remove any contaminants present (e.g. heavy metals and bacteria) prior to discharge. If the treated water can routinely attain appropriate contaminant guidelines (e.g. bacteria, metals and organic compounds), then discharge of these waters through the AES HBGS discharge vault would be acceptable if the treatment is designed to meet applicable standards that may be imposed by the Regional Board in the renewal of AES HBGS NPDES permit; and
- Although three of the four hypothesis involving the AES-HBGS have been rejected and the fourth demonstrated to not be significant in this study, there is evidence to suggest that the buoyant plume produced at the AES HBGS outfall may act to entrain non-AES sources of ocean water contamination (e.g. Talbert Marsh, Santa Ana river). Once entrained in the AES buoyant plume localized impacts to the surf zone around 9N can result. As a potential mechanism, the Thermal-Plume Onshore Transport mechanism (T-POT) warrants further investigation as discussed below.

T-POT Mechanism.

The T-POT mechanism has been proposed as a potential transport mechanism for pre-contaminated water from urban sources (local rivers and marshes for example) that interact with the AES HBGS thermal discharge to create a convergence of bacteria-contaminated water onto localized portions of Huntington State Beach. The mechanism has been based on observations taken during the thermal discharge dye study and previously obtained data (Jones and Svejkovsk, 2000), and would require analysis of the following variables for confirmation;

- Up-coast flow dye imagery;
- Modeling of both upcoast and down-coast convergence;
- Querying available data on up-coast/downcoast flow, microbial concentrations at potential source locations for urban run-off and at or in the vicinity of 9N;
- Calibration of model to available data; and
- Variation in model input parameters to determine impact of AES HBGS variables such as flow rate, temperature, direction of discharge, and location of discharge on thermal plume convergence.

In Addendum

Since the water quality investigation in the summer and fall of 2002, the City of Huntington Beach has conducted a source investigation to identify and address possible sources of bacteria within Blackford's Ditch. Four drains into the ditch from the AES HBGS were identified and sealed by AES staff and the volume of water within the ditch has been significantly reduced. An additional source of water was identified at the bottom of the pump house (Lucas, 2003).

8 LIMITATIONS

This report has been prepared for the exclusive use of the California Energy Commission as it pertains to the investigation of the AES Huntington Beach Generating Station in Huntington Beach, California. Our services have been performed using that degree of care and skill ordinarily exercised under similar circumstances by reputable, qualified environmental consultants practicing in this or similar locations. No other warranty, either expressed or implied, is made as to the professional advice included in this report. These services were performed consistent with our agreement with our client.

Opinions and recommendations contained in this report apply to conditions existing when services were performed and are intended only for the client, purposes, locations, time frames, and project parameters indicated. We neither warrant the accuracy of information supplied by others nor the use of segregated portions of this report.

The purpose of a surf zone water quality study is to assess the impacts of the Huntington Beach Generating Station on local fresh and saline water quality. In performing such an investigation, it is understood that no investigation is thorough enough to describe all hydrogeologic, oceanographic and microbiological conditions of interest at a given location. If conditions have not been identified during the investigation, such a finding should not, therefore, be construed as a guarantee of the absence of such conditions at the Site, but rather as the result of the services performed within the scope, limitations, and cost of the work performed.

In regard to hydrogeologic, oceanographic and microbiological conditions, our professional opinions are based in part on interpretation of data from discrete sampling locations. It should be noted that actual conditions at unsampled locations may differ from those interpreted from sampled locations.

Oceanographic data (salinity and temperature profiles, and current speeds and directions) presented and used herein were gathered by others and this report assumes that the data were gathered correctly and presented accurately. Water column profiles for August 2002 were gathered by Komex personnel at the AES HBGS discharge point. One of the August 2002 profiles has been used in this analysis.

The historical salinity and temperature profile data were collected near the discharge site (but not specifically at the Site). However, the data is considered representative of local conditions. Current data (speed and direction) were collected at two stations offshore of the discharge

(1,200 feet and 3,600 feet further offshore, respectively). The data were collected with in-situ current meters (Eulerian reference frame) and include mean speed and direction. However, a histogram identifying frequencies of specific current speeds was not provided.

"Drogue" data (Lagrangian reference frame) was not supplied or reviewed. The information is required to determine current movements "downstream" of where the current meters were installed, without this information, downstream movement can only be concluded as being unidirectional.

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APPENDIX A BACTERIA IN COASTAL WATERS: A REVIEW

A number of studies concerning enteric (of the intestines) bacteria survival and enumeration in saline waters have been conducted. In the marine environment, bacteria simultaneously encounter a range of abiotic and biotic factors affecting both viability and culturability; however, some of the bacteria demonstrate preferential ability to adapt to the marine environment. Of the abiotic factors challenging bacterial survival, photo-degradation is the dominant process, previous growth history plays a major part in pre-adaptation of bacterial cells, and stationary phase cells are generally more resistant than exponential and M. Grismerlly growing ones. Bacteria may take up temporary residence attached to suspended sediment or other easily suspended organic matter in the water column. Biotic factors, especially predation, primarily by protozoa, also reduce enteric bacteria survival in seawater. Finally, identification of a small number of genes in *E. coli* (primarily rpoS) that, when mutated upon transfer to seawater, significantly affect seawater sensitivity and adaptability, suggests that other enteric bacteria may also have such capability.

Colony formation-based coliform die-off rates are often the main parameter used to characterize the bacterial responses under a variety of biotic and abiotic test conditions. In one of the first extensive reviews, Grimes *et al.* (1986) hypothesized that enteric pathogens may survive for long periods of time in seawater.

Here, the recent review of microbiological factors affecting enteric bacteria survival in seawater by Rozen and Belkin (2001) is updated and expanded. Following discussion of colony formation and other modes of viability testing upon seawater exposure, their detection and possible transport in coastal environments is considered, the main biotic and abiotic factors reported to affect their sensitivity and survival in the sea is addressed, followed by a brief summary of the relatively little available information concerning molecular control of these effects.

CULTURES, VIABILITY, AND COLONY FORMATION

Colony formation ability has been used as the main or only viability parameter in many evaluations of seawater contamination by wastewater. Roszak and Colwell (1986) reviewed survival strategies of microorganisms while Grimes *et al.* (1986) highlighted the limitations of this approach. The viable but not culturable (VBNC) concept was introduced to describe cells

that remain metabolically active but are unable to divide in or on nutritional media that normally support their growth. The VBNC state was demonstrated for many enteric bacterial species during seawater incubation in the dark (Barcina *et al.,* 1997; Joux *et al.,* 1997; Pommepuy *et al.,* 1996; Davies *et al.,* 1995; Caro *et al.,* 1999; Troussellier *et al.,* 1998 and Grimes and Colwell, 1986) and in the light (Gourmelon *et al.,* 1994; Davies and Evison, 1991). Gauthier (2000), in a review of the environmental parameters associated with the VBNC state, discussed the effects of temperature, radiation, osmolarity, and organic matter concentration as possible triggers for the VBNC state and noted that, as in other cases, the multi-stress situation in seawater, along with the cells' previous history, plays an important role in their possible transition to VBNC status.

Since marine bacterial pollution is routinely monitored using culturing methods, findings that enteric VBNC bacteria in seawater can be resuscitated (Roth *et al.* 1988; Roszak *et al.,* 1984) or retain their pathogenicity (Pommepuy *et al.,* 1996a) are significant. Other studies, however, demonstrated loss of pathogenicity concomitantly with culturability (Caro *et al.,* 1999) or failed to resuscitate VBNC cells of several species using various methods (Bognosian *et al.,* 1996; Bognosian *et al.,* 1998). Although the latter two reports appear to be a minority, according to their authors they cast doubt on validity of the VBNC concept.

Interpretation of seawater viability studies depends largely on the diagnostic methodologies employed. Results are also influenced by the specific conditions during and even before the actual viability/pathogenicity experiments.

The ability to form colonies on solid media is generally considered to be lost first by enteric bacterial cells on exposure to seawater. Bordalo (1994) found no difference in fecal coliform recovery on two selective media: for water ranging from polluted seawater to unpolluted freshwater. Other studies, however, suggest otherwise (Davies *et al.,* 1995; Troussellier *et al.,* 1998; Garcia-Lara *et al.,* 1993; Fiksdal *et al.,* 1989; Galdiero *et al.,* 1994; Smith *et al.,* 1994), or in some cases (Barcina *et al.,* 1997; Joux *et al.,* 1997; Troussellier *et al.,* 1998; Bognosian *et al.,* 1996; Bognosian *et al.,* 1998) the viability represented as direct viable count (DVC) is lost. Total counts (TC) remain almost constant during all reported experiments (Barcina *et al.,* 1997; Joux *et al.,* 1997; Davies *et al.,* 1995; Troussellier *et al.,* 1998; Bognosian *et al.,* 1996; Bognosian *et al.,* 1998; Garcia-Lara *et al.,* 1993; Fiksdal *et al.,* 1989; Smith *et al.,* 1994).

Long incubation times (7 to 300 days) in seawater result in additional changes in cell viability states. Byrd and Colwell (1993) examined the survival pattern and plasmid maintenance of *E. coli* strains EK3C, H10407, and 34309 in an artificial seawater microcosm and found that a

portion of cells maintained a culturable phase for at least three years. Along with retaining culturability, that portion of the cell population also maintained their indigenous plasmids during the three-year period.

Garcia-Lara *et al.* (1993) reported that *E. coli* 536, previously grown at 30°C in a minimal medium, lost colony forming ability when transferred to sterile seawater (20°C) for 30 days, but maintained TC Cellular DNA concentrations and [^{3}H]-thymidine incorporation rates increased during the first five and 13 days, respectively, following which both parameters stabilized. Following incubation of *Salmonella typhimurium* cells in sterile, dark artificial seawater (20°C) for 19 days, Joux *et al.* (1997) and Trousselier *et al.* (1998) found that CFU decreased first, and cells showing metabolic activity in the presence of nutrients decreased in concentration to an undetectable level in four days. Respiration (with no nutrient addition) and potential respiration (after nutrient addition), both assayed by 5-cyano-2,3-ditolyltetrazolium chloride (CTC) reduction, decreased by three logs within ten and 13 days, respectively. The other three parameters measured – cellular integrity (TC), genomic integrity, and membrane permeability – remained constant during the entire experiment.

Bogosian *et al.* (1998) obtained somewhat different results in a long-term experiment (294 days) in which decay of *E. coli, K. pneumoniae, Enterococcus faecalis, Enterobacter aerogenes*, and *Salmonella choleraesuis* in artificial seawater microcosms (20°C) was monitored. CFU plate counts and DVC declined gradually and in parallel, while TC remained constant during the entire experiment, indicating cell membrane integrity, while respiratory activity was affected to differing extents between each species. In similar experiments conducted for 54 to 56 days at near-freezing temperatures by Smith *et al.* (1994) in Antarctica, stationary phase *E. coli, S. typhimurium, Yersinia enterocolitica,* and *E. faecalis* in diffusion chambers (-1.8°C) retained stable TC, but CFU, CTC reduction, and DVC decreased; the decay was gradual for the first three strains and rapid for *E. faecalis* (CTC reduction capability loss within ten days). Exponential phase *S. typhimurium* in sterilized seawater (22°C) during a 32-day period exhibited less than a ten-fold decrease in all cell count parameters studied – TC, DVC, and CFU (Galdiereo *et al.,* 1994). Cellular carbohydrate, lipid, protein, and DNA concentrations gradually decreased 75% to 95% at day 32, while RNA content did not decrease significantly. Hydrophobicity and adherence to eukaryotic cells decreased with seawater incubation time, while susceptibility to phagocytosis increased. All of the above correlated with the virulence of *S. typhimurium* as tested on mice; mortality of infected animals (at day seven) decreased from 80% on day one to 30% by the 16th day.

The ability to form colonies on solid media is largely dependent on the medium used. Morinigo *et al.* (1990) defined sublethal cellular injury as the loss of ability to form colonies on selective media, while retaining it on less selective ones. Dawe and Penrose (1978) found that CFUs enumerated on seawater-based nutrient agar remained constant for six days, while on standard Levine EMB agar counts declined to zero by the fifth day for marine *E. coli* isolates from a dialysis bag submerged in seawater. The same protective effect of the recovery medium salinity was observed by Gauthier *et al.* (1987), who found that the greatest recovery rate of *E. coli* K12 incubated in seawater was on a nonselective medium (nutrient agar) supplemented with sodium chloride (NaCl) (15 g/L).

Assuming that enzymatic activities in VBNC cells are detectable, disparities may be expected between colony-based counts and coliform enumeration assays based on enzymatic activities; resulting in potential discrepancies in bacterial monitoring. In studies monitoring sewage pollution of marine waters, such disparities were absent (Apte and Bailey, 1994; Fiksdal *et al.,* 1994); however, when comparing enzyme activity of pure cultures to their colony-forming ability, such discrepancies are indeed recorded (Fiksdal *et al.,* 1989; Davies *et al.,* 1995; Pommepuy *et al.,* 1996b). The latter two studies described D-galactosidase activity in seawater even when no culturable cells were detected. For example, culturability of an environmental *E. coli* isolate decreased to zero after 60 days while viability decreased by one to two logs during the same period. During the 85 days in seawater, D-Galactosidase activity, after an initial increase, remained constant for the duration of the experiment even when the number of culturable cells reached zero (Davies *et al.,* 1995). Using *E. coli* H10407, Pommepuy *et al.* (1996b) also observed a decrease in culturable cells while enzymatic D-Galactosidase activity remained constant. Repeating the experiment under visible light exposure reduced enzyme activity, though it was still detectable when no colony-forming cells remained. Using the same *E. coli* strain in seawater, Fiksdal *et al.* (1989) measured increased 4-methylumbelliferyl heptanoate hydrolase activity while during the first 16 days CFUs decreased by two to three logs. Similarly, *E. coli* strains JM83 and JM101 were shown to maintain plasmids (pBR322 and pUC8, respectively) at a high copy number for at least 21 days in artificial seawater microcosms, even though both strains lost their ability to form colonies on a solid medium (Gameson and Gould, 1975).

Resuscitation of VBNC cells has been demonstrated using several methods. Restoration of colony-forming ability in about 80% of osmotically stressed *E. coli* CA8000 cells by betaine (2 mM) was achieved within two hours (Roth *et al.,* 1988). Colony-forming ability could still be restored after 24 hours in NaCl solution, even in the presence of the protein synthesis inhibitor chloramphenicol. This latter observation indicates that the betaine-driven increase in CFU was

not due to cryptic growth of culturable cells. Roszak *et al.* (1984) employed a different resuscitation strategy for *Salmonella enteritidis* C1 cells revived four days after the nonculturable state was achieved, by a 25-hour incubation in full strength veal infusion broth followed by colony formation on veal infusion agar.

The question whether *E. coli* can actually enter VNBC state was critically examined by Bogosian *et al.* (1996) using stationary phase strain W3110 cells in water and soil. In sterile artificial seawater, total counts held constant during the 56 days of the experiment; simultaneously, viability decreased by three or five logs (at 20°C and 37°C, respectively) regardless of the parameter monitored (CFU, DVC, or growth in liquid medium). Attempts to resuscitate these VBNC cells by various means, including incubation in different media, a shift to a low temperature, and an osmotic adaptation, did not succeed. In a subsequent study, Bogosian *et al.* (1998) attempted resuscitation of *E. coli, K. pneumoniae, E. faecalis, E. aerogenes,* and *S. choleraesuis* cells incubated in artificial seawater for nearly 300 days by nutrient addition or a temperature shift (20°C to 4°C or to 37°C) both in the presence and in the absence of culturable cells. At all time points tested, only cells of the culturable strains could be recovered.

In their study of the potential pathogenicity of stationary phase S*. typhimurium* C52 VBNC (driven by sterile artificial seawater and UV-C stress) cells, Caro *et al.* (1999) claimed loss of both culturability and pathogenicity (tested in a mouse model), independent of viability level. Cell viability as determined by respiratory activity (CTC reduction), cytoplasmic membrane (fluorescent staining), and genomic integrity (TC) remained unchanged, but metabolic activity (DVC) declined together with culturability and could not be distinguished from the latter. In contrast, Pommepuy *et al.* (1996) demonstrated pathogenic effects of *E. coli* H10407 cells induced to the VBNC state by a combined seawater and light exposure. As DVC counts were almost constant during the experiments, and no culturable cells could be detected, it was interpreted that the entire population produced the enterotoxin rather than a few active cells.

It appears that in general, the hostile situation encountered by enteric bacteria in seawater promotes loss of colony-forming ability while maintaining different aspects of viability. That is, their fate can thus be considered a special case of the VBNC phenomenon (Gauthier, 2000). While there are several stress factors in the marine environment that may be more important, there are similarities between this situation and cell responses to starvation, often a major factor forcing entry into the VBNC condition.

ENTERIC BACTERIA DETECTION IN COASTAL WATERS

Enteric bacteria have been detected in a variety of ocean beaches around the world. De Donno *et al.* (1994) evaluated the influence of the number of swimmers on microbiological quality of three different beaches in Salento, Italy, an area subject to minimal fecal pollution. Daily and hourly variations in microbial densities were observed, and count fluctuations of total fungi, Enterococcus, fecal coliforms, and *E. coli* were significantly correlated to the numbers of swimmers. In a survey of the occurrence of enteric and non-enteric indicators in seawater along the beaches of southern Greece during the summer, Papapetropoulou and Rodopoulou (1994) detected total coliforms, fecal coliforms, fecal streptococci, *Staphylococcus aureus*, *Pseudomonas aeruginosa, Aeromonas hydrophila*, total fungi, and *Candida albicans* in 78.5%, 71.6%, 86.8%, 6.8%, 12.4%, 5.6%, 89.4%, and 3.7% of the 265 samples, respectively. Fecal streptococci were recovered in 15.1% and 19.6%, respectively, of the samples in which total and fecal coliforms were not recovered. Similarly, in approximately 1 to 2% of the samples, *Staphylococcus aureus, Pseudomonas aeruginosa, Aeromonas hydrophila,* and *Candida albicans* were isolated when both total and fecal coliforms were absent. These supplementary bacterial water quality indicators suggest that the inclusion of fecal streptococci and total fungi monitoring with total and fecal coliform sampling may provide greater protection of public health in popular marine beaches. Similarly, in developing a microbiological surveillance program for beaches near Durban, South Africa, to provide an objective assessment of changes in the local seawater quality before and after the commissioning of two submarine outfalls, Rathbone *et al.* (1998) included measurement of *E. coli* I, helminthic parasite ova, pathogenic staphylococci, salmonellae, shigellae counts, and salinity. They found that alterations in the seawater quality were invariably a consequence of onshore changes or meteorological events and that measuring more than one water quality indicator, plus salinity as a physical parameter for assessing dilution or impairment of seawater, was quite valuable.

Many studies have considered the effects of near shore land use or urbanization processes on estuarine enteric bacteria levels. Other related environmental effects may play a role in enteric bacteria survival in seawater. Based on presumptive *E. coli* counts determined in 149 seawater samples taken from three locations at distances of 1.9, 2.4, and 4.3 km (1.2, 1.5 and 2.7 miles) from a sewage outfall and correlated with data on wind speed and direction in the three hours prior to sample collection, Smith *et al.* (1999) found that numbers of presumptive *E. coli* present in a sample were significantly greater when the sample site lay downwind of the outfall. Wind speed was shown to have an influence on the numbers of presumptive *E. coli* only when the sample site was downwind of the outfall. These data demonstrate that wind speed and

direction at the time of sampling significantly influence numbers of presumptive *E. coli* detected in any seawater sample.

When evaluating bacteriological conditions of the Fortaleza coastal region (Brazil), Melo *et al.* (1997) found that the spatial distribution for Salmonella suggested existence of an east to west shoreline sea current. Using a multivariate (principal component) analysis on total and fecal coliforms; fecal streptococci; nickel(II), zinc(II), lead(II), cadmium(II), copper(II) and chromium(VI) concentrations; nitrates; phosphates; DO; pH; and conductivity from 919 water samples corresponding to 52 sampling points along the coast of Valencia, Spain, Morales *et al.* (1999) established the sources and types of marine contamination, identifying urban runoff and waste disposal as the source of fecal contamination.

Al-Muzaini *et al* (1999) also evaluated several metal, nitrate, sulfate, and phosphate species together with biological oxygen demand (BOD), chemical oxygen demand (COD), and coliforms near a major sewage outlet located close to Shuwaikh Harbor (Kuwait) and, not surprisingly, found lower levels of chemical pollutants and fecal counts at high tide as compared to low tide. Pires-Coelho *et al.* (1999) also correlated lower total coliform counts with lower marine salinity, but higher counts at higher turbidity during high and low tides. In fact, turbidity peaks from stormy weather were accompanied by peaks of microbial counts, indicating that the contamination is normally deposited at the marine sediment rather than in the water column.

Mallin *et al.* (2000) analyzed the abundance and distribution of fecal coliforms and *E. coli* across a series of five estuarine watersheds, each of which differed in both the amount and type of anthropogenic development. Within all creeks, there was a spatial pattern of decreasing enteric bacteria away from upstream areas, and both fecal coliform and *E. coli* abundance were inversely correlated with salinity while turbidity was positively correlated with enteric bacterial abundance. Fecal coliform abundance was significantly correlated with watershed population, and even more strongly correlated with the percentage of developed land within the watershed. The most important anthropogenic factor associated with fecal coliform abundance, however, was percentage watershed-impervious surface coverage (*i.e.* roofs, roads, driveways, sidewalks, and parking lots). These surfaces serve to concentrate and convey storm-water-borne pollutants to downstream receiving waters. Impervious surface area alone explained 95% of the variability in average estuarine fecal coliform counts.

In contrast, Grant *et al.* (2001) found that the dominant source of Enterococcus bacteria in southern California beaches (including the study area) was not urban runoff and wastewater

discharges, but appeared to be of avian origin in a recently restored marsh (Talbert Marsh). Bacterial counts at all measurement locations exhibited a diurnal effect readily correlated with solar radiation. However, in simple correlation analysis of their data, high bacterial counts could be also be associated with high turbidity but were independent of measured salinity near shore. A more in-depth power spectrum analysis of their water level, flow velocity, solar radiation, salinity, water temperature, bird count, and Enterococcus count data may provide some insight into the time lags associated with some of the near shore processes affecting surf zone bacterial levels.

In subsequent studies, Boehm *et al.* (2002a & 2002b) found that cross-shelf currents (internal tides) were capable of transporting sewage outfall waters back into the surf zone at water depths on the order of ten m (33 feet) on both semi-diurnal and diurnal frequencies, thereby not precluding sewage outfall as a possible contamination source. Conversely, they identify that a complex of seasonal and El Niño type factors affect coliform level variability as modulated by phases of the moon effects on tidal flows such that there is a five to eight year cycle. Such studies suggest that near shore processes are complex and that identification of a single source of fecal contamination is unlikely.

Development of more accurate and rapid bacterial contamination tests is needed to provide adequate public warnings. Zaccone *et al.* (1995) showed that the rapid microscopic indirect immunofluorescent method had high specificity for entero-pathogenic *E. coli,* while they later (Caruso *et al.,* 2000) found that determination of *E. coli* populations in marine waters through this technique was equivalent to the conventional count on an agar medium, with a detection limit of 0.01 cells/100 ml for immunofluorescence. Such rapid measurement techniques may be more useful than standard plating culture techniques in devising public health warnings. By the same token, care must be taken in developing rapid testing methods. Pisciotta *et al.* (2002) noted that while the Colilert-18 system for enumeration of total coliforms and *E. coli* is approved by the USEPA for use in drinking water analysis and the technique is used by various agencies and research studies, they found that its use resulted in *E. coli* numbers frequently exceeding fecal coliform counts (by membrane filtration) by one to three orders of magnitude in Florida seawater samples. In contrast, when comparing results from 22 southern California laboratories, McGee *et al.* (2001) found no significant differences among measurement methods of *E. coli*, fecal coliforms and total coliforms.

Of perhaps greater concern is that many pathogenic bacteria and viruses demonstrate greater survival in seawater than *E. coli*. Using comparative survival tests of fecal coliforms, fecal streptococci, Salmonella spp., and *S. aureus* grown in seawater, Gabutti *et al.* (2000) noted that

survival of *S. aureus* was greater than fecal streptococcus, which in turn was greater than Salmonella spp. and fecal coliforms, and suggested that *S. aureus* could be an effective indicator of human pollution. Wait and Sobsey (2001) measured lower survival of *E. coli* as compared to *Salmonella typhi, Shigella sonnei*, poliovirus type 1, and a parvovirus cells in lab and in situ seawater irrespective of water temperatures. Cell survival in the laboratory was greater for all compared to those in situ in the Atlantic Ocean. Noble and Fuhrman (2001) used molecular-based assays for the detection of enteroviruses by reverse transcriptase polymerase chain reaction (RT-PCR) on 50 coastal seawater samples taken from freshwater outlets in Santa Monica Bay and popular sandy beaches during a six-year period and compared it with indicator bacteria counts. They found that ultrafiltration concentration methods and RT-PCR protocol could be used to consistently detect enteroviruses. Moreover, there was no significant correlation between the presence of enteroviruses and individual counts of total coliforms, fecal coliforms, or Enterococcus; however, there was a significant correlation to the combined set of all three indicators, a standard recently adopted in California.

ABIOTIC BACTERIAL MORTALITY FACTORS

LIGHT EFFECTS

Enteric bacteria survival in the sea is greatly affected by UV and visible light and light is considered to be the single most important factor in bacterial die-off at sea (Gameson and Gould, 1975; Chamberlin and Mitchell, 1978; Fujioka *et al.,* 1981; Sinton *et al.,* 1994). Though UV effects seem restricted to shallow depths (Sinton *et al.,* 1994), radiation wavelength has an important depth-dependent effect (Davies-Colley *et al.,* 1994; Sinton *et al.,* 1999).

Several studies have shown the photo-toxicity effects on *E. coli* in seawater result in a rapid decrease of colony-forming ability (*e.g.* Troussellier *et al.,* 1998; Davies and Evison, 1991; Fujioka *et al.,* 1981; Basrcina *et al.,* 1990). For example, Gourmelon *et al.* (1994) estimated the effect of visible light on *E. coli* H10407 in seawater by loss of colony-forming ability. Illumination of *E. coli* suspended in oligotrophic seawater with visible light caused a sharp decrease of culturable bacteria, which turned into a VBNC state. In the seawater microcosm, *E. coli* exhibited weak metabolic activity, based on ^{3}H methyl-thymidine incorporation in the cell, but this activity was not significantly affected by visible light and did not involve detectable oxidation of lipid membranes as evaluated by gas chromatography analysis of fatty acids. A decrease of the phototoxic effect was observed when *E. coli* were exposed to visible light under anaerobic conditions. Pommepuy *et al.* (1996) showed that *E. coli* responded to the estuarine diurnal solar cycle by entering the VBNC state upon exposure to sunlight. That is, DVC remained stable

without significant change, but *E. coli* (H10407) cells remained fully culturable only when exposed to seawater in control chambers in the dark (*i.e.* without solar irradiation). UV-B (280 to 320 nanometers [nm]) wavelengths of the solar spectrum are the most bactericidal, causing direct photo-biological DNA damage (Calkins and Barcelo, 1982). At longer wavelengths, photochemical mechanisms become more important, usually acting through photo-sensitizers and tending to be more injurious in the presence of oxygen (Sinton *et al.,* 1994).

Greater sunlight exposure is required to inactivate Enterococcus as compared to fecal coliforms in seawater (Sinton *et al.,* 1994; Davies-Colley *et al.,* 1994; Sinton *et al.,* 1999) and Sinton *et al.* (1999) found the fecal coliforms to be more sensitive to light inactivation than fecal bacteriophages (somatic coliphages, F- DNA phages, F- RNA phages, and *Bacillus fragilis* phages). Light was also demonstrated to exert a significant negative effect on culturability in an in situ experiment, where natural sewage populations of *E. coli* and Enterococcus were placed 1.0 to 1.5 m deep in seawater for seven days. The effect was pronounced when batch culture cells were used, but not in diffusion chambers (Lessard and Sieburth, 1983).

In other studies, exposure of *E. coli* and *S. typhimurium* to light (~ 1 MJ m^{-2} h^{-1}) in seawater for four and ten hours, respectively, did not affect TC, while DVC and CFU in both strains decreased by four to five logs (Davies and Evison, 1991). Pommepuy *et al.* (1996a) demonstrated that *E. coli* cells showed stable DVC and TC counts following a 26-hour exposure to natural sunlight in seawater, but CFUs decreased from 2×10^6 /ml to an undetectable level. Gourmelon (1995) obtained somewhat different effects for *E. coli* H10407 under artificial visible light; concentrations were unaffected when measured as TC, while DVC and CFU exhibited approximately 1.5 to four log reduction, respectively, within a 40 hour experiment. Troussellier *et al.* (1998) presented similar results for *E. coli* exposed to visible light (125 W m^{-2}) for the same duration; that is, no decrease in TC, approximately one log decrease in DVC, and a six log decrease in CFU, whereas for non-illuminated bacteria, only a 1.5 log decrease in the CFU was measured, with no effect on either TC or DVC. Kapuscinski and Mitchell (1981) showed sublethal injuries to an *E. coli* isolate incubated in autoclaved filtered seawater exposed to sunlight; injuries were apparent during the recovery stage on complete or minimal media, and were not observed in the dark. Addition of either pyruvate or catalase to the minimal medium restored colony-forming ability to that of complete medium, implying at least a partial involvement of peroxide damage.

The role of reactive oxygen species in photo-toxicity was later studied using *E. coli* mutants, defective in specific genes involved in antioxidant defense systems (Gourmelon *et al.,* 1997); colony counts were directly related to presence of functional rpoS genes when exposed to

artificial light (0.9 mmol photons m^2/s) for 50 to 100 hours. RpoS is a factor involved in various environmental stresses, and its presence has also been shown to have a protective effect for stationary phase *E. coli* in seawater (Munro *et al.,* 1994 and 1995) and its function is considered in greater detail below. The rpoS light-protective effect was evident only in stationary phase cells, and a mutation in the rpoS gene caused a CFU decrease of about five logs as compared to wild-types. Stationary phase mutants, defective in genes involved in resistance to hydrogen peroxide (katE katG (catalases), dps (DNA binding protein), and katE katG dps) were all more sensitive to light (one to two logs) than their corresponding wild-types. In exponentially growing cells, Gourmelon *et al.* (1997) found that a mutation in oxyR, the regulatory gene of the adaptive response to H_2O_2, led to an increased sensitivity to light (~one log), further suggesting that the deleterious effects may be associated with H_2O_2 production. However, the increased sensitivity of the rpoS mutant suggested that rpoS-dependent protection against deleterious effects may be independent of H_2O_2 removal. Possible involvement of superoxide radicals was indicated by an enhanced sensitivity of the double superoxide dismutase mutation (sodA sodB); individually, each of these mutations had no effect. Other mutants not affected by light were the double mutants otsA otsB (trehalose synthesis), fpg uvrA, recA (DNA repair), xthA (exonuclease III), and katE. Interestingly, a mutation in hemA (glutamyl-tRNA reductase), involved in δ-aminolevulinic acid formation from glutamate, had a protective effect (Gourmelon, 1995).

Aside from genetic mutations, hazardous solar radiation effects may be mitigated by the presence of dissolved organic material (Calkins, 1982), chlorophyll, and particulate matter (Baker and Smith, 1982; Jerlov, 1976). Such effects are more pronounced in relatively eutrophic coastal and estuarine areas, and are particularly relevant for UV-B radiation due to selective absorption of shorter wavelengths such that the bacteria in such areas are primarily exposed to visible light (400 to 775 nm) and, to a lesser extent, UV-A (320 to 400 nm) radiation. Conversely, increased salinity enhances the bactericidal effects of visible light, suggesting a synergistic effect of the combined stress factors. It was proposed (Gourmelon, 1995; Gourmelon *et al.,* 1997) that the salinity-driven sensitization is mediated by endogenous components, such as protoporphyrin or ubiquinone biosynthesis intermediates.

SALINITY EFFECTS

General salinity or osmolarity effects on fecal bacteria are well known and can be enhanced by sunlight. Troussellier *et al.* (1998) considered the interactive effects of salinity (sea salts, 3.7% w/w), light (sunlight, approximately 300 W m^{-2}), and presence of organic matter (100 mg/L glucose) and found that salinity increased *E. coli* sensitivity (CFU) only in the presence of light,

regardless of the presence of organic matter. Survival of *E. coli* in seawater/distilled water mixtures (0, 25, 50, 75, and 100% seawater) for 48 hours showed optimal survival (74%) in 25% seawater, as compared to 60% and 8% survival, respectively, in distilled water and 100% seawater (Carlucci and Pramer, 1960b). Sublethal salinity effects were also reported by Anderson *et al.* (1979), who monitored the survival of an *E. coli* isolate for eight days in seawater salinities of 1, 1.5, 2.5, and 3%; decreasing salinity was accompanied by increasing survival. Similar results were obtained when NaCl solutions were used instead of seawater. Smith *et al.* (1999) demonstrated an inverse correlation between salinity and log presumptive *E. coli* numbers, suggesting that when released into the sea, enteric bacteria are subjected to an immediate osmotic upward shock, and their ability to overcome this by means of several osmo-regulatory systems could largely influence their subsequent survival (Gauthier *et al.*, 1987; Munro *et al.*, 1989].

Flatau *et al.* (1994) investigated the effects of osmotic shock on uptake and release of carbohydrates and amino acids by *E. coli* cells grown in non-salted medium with salted buffer or seawater. Upward-shocked cells could restore, at least to some extent, their ability to accumulate nutritive substrates and not release them into the medium upon an osmotic upshift; bacterial cells accumulate or synthesize specific osmoprotectant molecules in order to equalize osmotic pressure and avoid drastic loss of water from the cytoplasm (Csonka and Epstein, 1996). The accumulation or synthesis of such molecules (trehalose, glycine betaine, glutamic acid) in *Salmonella manhattan* and *S. typhimurium* in estuarine waters (3.5% salinity, 20°C) was detected. *S. manhattan* accumulated trehalose and an unidentified substance, whereas *S. typhimurium* accumulated mostly glycine betaine (Troussellier *et al.*, 1998). Trehalose synthesis was also observed for wastewater-borne *S. manhattan* in oligotrophic seawater (Dupray and Derien, 1995). It was also shown that an osmotic upshock combined with nutrient deprivation inhibited several transmembrane transport systems within 60 minutes in different *E. coli* strains. During a much longer incubation (30 days), Flatau *et al.* (1994) found different accumulation patterns for histidine (constant accumulation) and maltose (increased during the first two weeks and than slowly decreased). Glutamate added to filter-sterilized seawater was shown to enhance *E. coli* MC4100 culturability. The effect appeared to be logarithmically correlated to glutamate concentration; glycine betaine uptake and its protective effect were both enhanced in the presence of glutamate (Gauthier *et al.*, 1993).

The essential role of osmo-regulatory mechanisms in enteric bacterial survival in seawater was demonstrated in several studies. Munro *et al.* (1987 and 1989) and Gauthier *et al.* (1987) found that cells pre-adapted to high osmolarity are highly resistant to seawater, but this pre-adaptation depended upon the media used. The induced systems shown to be of value in

conferring a protective effect for growing *E. coli* MC4100 were potassium transport, glycine betaine synthesis or transport, trehalose synthesis (Munro *et al.* 1989), and glutamate accumulation (Gauthier *et al.,* 1991). Survival of non-adapted *E. coli* cells was not affected by impaired potassium transport or glycine betaine uptake, while trehalose synthesis ability was apparently important. The protective advantage of glycine betaine in nutrient-free seawater seems to vary (Munro *et al.,* 1989; Gauthier and Le Rudulier, 1990). Resistance of *E. coli* M9 grown in saline media (0.5 M NaCl) to seawater was suppressed by an osmotic downshock that resulted in a loss of several osmolytes previously accumulated (Gauthier *et al.,* 1991). An osmotic downshock (distilled water or wastewater) increased loss of culturability (Gauthier *et al.,* 1991; Combarro *et al.,* 1992).

PH EFFECTS

Seawater pH typically ranges between 7.5 and 8.5 and is influenced by temperature, pressure, and the photosynthetic and respiratory activities of microorganisms (Harvey, 1955). An acidic pH (~5.0) was found to be most favorable for *E. coli* survival (in the 5.0–9.0 range) in both seawater and NaCl solutions, and sensitivity increased with the increase in pH (Carlucci and Pramer, 1960b). They concluded that seawater pH, normally about 8.0, contributes to the deleterious effects on *E. coli* survival.

NUTRIENT DEPRIVATION EFFECTS

It is often overlooked that with sufficient nutrient levels, *E. coli* grow in seawater nearly as well as in rich laboratory media. Jannasch (1968) demonstrated that *E. coli* competed successfully with five marine bacterial isolates in nutrient-enriched seawater. Lopez-Torres *et al.* (1988), in their study of *K. pneumoniae* and *E. coli* in membrane diffusion chambers located at coastal areas in Puerto Rico, showed increased survival of *E. coli* and to a lesser extent of *K. pneumoniae* (respiring cells) in a site receiving rum distillery effluents. The density of *E. coli* declined immediately after the effluent discharge stopped, suggesting that the organic load improved their survival, this was already shown by Carlucci and Pramer (1960b), who found that *E. coli* survival improved with increasing nutrient concentrations, both organic and inorganic. The effect of organics was pronounced when peptone and sewage volatile solids were added, but not glucose. The effect of sewage volatile solids was more pronounced in filter-sterilized seawater, indicating a possible competition for available nutrients with indigenous marine microflora.

Similar results were obtained by Troussellier *et al.* (1998); who found that when only nutrient deprivation (glucose) was imposed, a small energy charge decrease was observed but CFU and

transport abilities were maintained. Combined with the other stress factors, nutrient deprivation led to the inactivation of membrane transport and to a pronounced reduction in energy charge. However, outside of contaminated coastal zones, levels of inorganic and organic nutrients in seawater are dramatically lower than those of laboratory media or wastewaters such that bacteria released into the marine environment must contend with starvation conditions. Rozen and Belkin (2001) indicated that while the presence of nutrients actually allows *E. coli* to grow in seawater, their absence does not necessarily affect survival (colony formation) of non-growing cells. In fact, cells induced to grow in nutrient-enriched seawater were more susceptible to various mutations (kdpABC trkA, kdpABC trkE [potassium transport], nhaA [Na^+/H^+ antiporter], hns [histone-like protein], sodA sodB and rpoE [σE]] that had no effect in un-amended seawater as summarized in a comprehensive table of genetic mutation effects on *E. coli* in seawater.

TEMPERATURE EFFECTS

Low seawater temperatures are another obvious shock for microorganisms whose optimal growth occurs around 37°C or 98.6°F (Ingraham and Marr, 1996); however, optimal survival temperature is not necessarily the same as that for growth, and most reports indicate enhanced stability of *E. coli* at lower temperatures. Such results were obtained by Carlucci and Pramer (1960a) in a 48-hour experiment in natural seawater (5 to 40 °C), and by Vasoncelos and Swartz (1976) in a 6-day experiment in diffusion chambers exposed to temperatures ranging from 8.9 to 14.5 °C. A similar tendency was reported by Lessard and Sieburth (1983), who measured viability (CFU) of sewage populations of *E. coli* and Enterococcus in diffusion chambers from February to August at temperatures ranging from 0°C to 20°C. Though other environmental changes may have contributed to bacterial survival, survival was significantly greater at the lower temperatures. Short-term experiments (6-hour) measuring survival (TC, DVC, CFU) of *Salmonella montevideo* in filtered seawater exposed to visible light (1.03–1.12 MJ m^{-2} h^{-1}) at three different temperatures (5, 15, and 25°C) showed no changes in any parameters tested, though lack of differences among the three temperatures was possibly due to the short duration of the experiment (Davies and Evison, 1991).

HYDROSTATIC PRESSURE EFFECTS

Wastewater release into deep waters offshore was originally considered a solution for immediate public health hazards caused by disposal, resulting in several studies of fecal microorganism die-off rates in near shore conditions; however, little study of their resistance and fate under deep-sea conditions and possible re-circulation back into surf zones (*e.g.* Boehm

et al., 2002) has been conducted. Welch *et al.* (1993) described response of *E. coli* to elevated hydrostatic pressure at both the whole cell physiology and proteins levels. Water pressure increased to 546 atm dramatically inhibited *E. coli* W3110 growth measured as optical density, CFU, or TC (epifluorescence direct counts). A linear decrease in the rate of protein synthesis was observed with pressure increasing up to 1,092 atm, where it essentially ceased. Increased pressure also modulated synthesis of specific proteins; two-dimensional gels revealed induction of 55 proteins, 11 of which were shown to be heat-shock proteins and 4 were cold-shock proteins. Under simulated deep-sea conditions (4 °C and pressures up to 1,000 atm), Baross *et al*., (1975) found survival of pure cultures of four human enteric bacteria and a sewage bacterial population to be species-dependent. Following 12 days of incubation at hydrostatic pressures up to 1,000 atm, survival of *Clostridium perfringens, Vibrio parahaemolyticu,* and an aerobic sewage bacterial population was negatively affected by increasing pressure, while *E. coli* and *S. faecalis* exhibited greater survival rates at 250 and 500 atm pressure than at one or 1,000 atm. It should be noted that reported hydrostatic pressure effects (even at 1,000 atm) on bacterial survival are quite variable and apparently species-specific. Nonetheless, release of raw or partially treated sewage into deep waters does not guarantee reduction in the health risks inherent in wastewater-borne microorganisms.

BIOTIC FACTORS AFFECTING BACTERIAL SURVIVAL IN SEAWATER

Though poor in nutrient concentrations, the marine environment is nevertheless inhabited by relatively diverse biological populations influencing enteric bacteria survival in seawater. Indeed, observations (Gauthier *et al*., 1987; Gonzalez *et al*., 1992) of lower enteric bacteria survival rates in natural seawater as compared to sterile seawater suggest involvement of biological processes. Le Guyader *et al.* (1991) provided support for this hypothesis by measuring earlier and at faster declines of *E. coli* viability (CFU) in seawater and sediment during a 13-day period when indigenous seawater flora were present as compared to sterile conditions. Predation (Davies *et al*., 1995; Gonzalez *et al*., 1992; Greenberg, 1956; Mitchel and Nevo, 1965; Mitchel *et al*., 1967; Guelin *et al*., 1967; Mitchel and Morris, 1969; Enziinger and Cooper, 1976; Barcina *et al*., 1992), competition (Jannusch, 1968; LeGuyader *et al*., 1991; Greenberg, 1956; Mitchell, 1968) and bacteriophages (Guelin *et al*., 1967; Carlucci and Pramer, 1960) have been implicated in reducing enteric bacterial concentrations in seawater. Each of these processes is considered in more detail below.

PROTOZOA

Various studies indicate that the main predators of bacteria in the marine environment are protozoa; more recently, Hartke *et al.* (2002) identified zooflagellates as the primary bacteriovore. Differential predation of *E. coli* as compared to Enterococcus or streptococci is also common. Using filters with different pore sizes and antibiotics to suppress indigenous bacterial activity, it was shown that bacterial competition, antagonism, and even bacterial predation were relatively unimportant in coliform removal. It has also been shown that reduction in *E. coli* populations paralleled an increase in the number of protozoa. In the same vein, survival of fecal coliforms was significantly improved in the presence of cyclohexamide, a compound expected to inhibit eukaryotic organisms. Interestingly, *Clostridium perfringens* and fecal streptococci were not affected by the addition of this inhibitor, which could be explained either by differential resistance of protozoan species to cyclohexamide or by selective predation on the different bacterial species. Gonzalez *et al.* (1992) enumerated *E. coli* and *E. faecalis* for five days in natural and filtered (0.2 micrometers [um]) seawater and found that both CFU and DVC decreased significantly (three to five logs) in the presence of natural microbiota, an effect that was attributed to protist activity.

Mitchell *et al.* (1967) demonstrated that seawater samples plated on *E. coli* lawns on solid agar resulted in bacterial colonies surrounded by clear zones attributed to bacteria utilizing the cell wall of *E. coli* as a carbon source. Also observed were plaques, possibly caused by obligate parasites such as Bellovibrio. Mitchell and Nevo (1965) isolated from seawater a Pseudomonas spp. capable of utilizing Flavobacterium capsular polysaccharide as its sole carbon source. They also isolated bacteria capable of growth on capsules of Azotobacter, Rhizobium, Arthrobacter, and the cell wall of *E. coli*. Activity of the isolated Pseudomonas against living cells of Arthrobacter and *E. coli* was tested by joint growth in artificial seawater with 0.1% (w/w) peptone. The authors concluded that growth of both terrestrial bacteria was markedly suppressed by the marine bacterium, though this was based on optical density rather than on a more accurate enumerating method.

COMPETITIVE INHIBITION

The role of competition with marine bacteria has been investigated historically (Jannusch, 1968; Mitchell *et al.*, 1967). Experiments measuring *E. coli* survival in autoclaved seawater with different sizes of marine microbial inocula showed a clear effect on *E. coli* CFU counts. In the presence of marine bacteria at a concentration of 10^7 ml^{-1}, *E. coli* population decreased by about four logs during a seven-day period, while in autoclaved seawater almost no decrease was

detected (Mitchell *et al*., 1967). Furthermore, when a mixed marine *E. coli and* bacteria culture was reinoculated with *E. coli* after 7 days, its elimination from the medium proceeded rapidly with no lag phase. The enhanced removal rates suggested existence of microflora parasitic or lytic to *E. coli*. Jannusch (1968) selected and isolated organisms defined successful competitors under various dilution rates or different limiting nutrient concentrations in chemostat experiments and found *E. coli* to be a successful competitor in rich media but a very poor one under the low nutrient concentrations characterizing natural seawater.

COLIPHAGE

Coliphages active against *E. coli* have been detected in sewage-contaminated marine waters (Sinton *et al*., 1999; Guelin *et al*., 1967; Carlucci and Pramer, 1990; Sogaard, 1983; Paul *et al*., 1993 and 1997) including *Aerobacter aerogenes* and *Serratia marinorubra*. Presence of coliphages has been positively correlated with fecal phages, enteric viruses, and other pathogens (Sogaard, 1983; Paul *et al*., 1997; Legnani *et al*., 1998; Marino *et al*., 1995; Borrego, 1987; Dutka, 1987). Nevertheless, the presence of enteric bacterial infectious phages does not necessarily indicate their actual activity in removing coliforms from marine water. Carlucci and Pramer (1960) have shown that bacteriophages were effective in reducing *E. coli* population sizes only under nutrient-rich conditions, suggesting a very minor role for bacteriophages, if any, under natural conditions. This was corroborated by Penon *et al*. (1991), who found no significant deleterious effect of the seawater fraction passing 0.2 μm filters (containing bacteriophages, but not bacteria or protozoa) on the survival of fecal bacteria in seawater. The 0.2 to 2-um fraction, in contrast, had a significant effect.

Very early studies suggested that deleterious effects of seawater on enteric bacteria, not explicable by other factors known at the time, were caused by antibiotics produced by microorganisms (Greenberg, 1956; Rosenfeld and ZoBell, 1947; Vaccaro *et al*., 1950; Richou *et al*., 1955), or negative effects of an algal toxin (Carlucci and Pramer, 1960). However, all reported attempts to find such an antibiotic effect of seawater on enteric bacteria yielded no evidence that such compounds are produced under natural seawater conditions.

BACTERIAL SURVIVAL IN MARINE SEDIMENT

Occurrence of coliforms or Enterococcus in estuarine and coastal sediments are reported to be ten to 100 fold greater than in the water column above (Gerba and McLeod, 1976; Shiaris *et al*., 1987). Such reports have been the impetus for detailed studies of how these organisms indeed survive better in sediments (Gerba and McLeod, 1976; Hood and Ness, 1982) and to reveal the factors involved in the sediment protective effect (Gerba and McLeod, 1976; Shiaris *et al*., 1987;

Van Donsel and Geldreich, 1971; King, 1984). Longer survival time of *E. coli* in sediments was attributed to the higher organic matter content of the sediment. One of the compounds shown to be present was glycine betaine (King, 1984); indeed, marine *E. coli* isolates were shown to accumulate glycine betaine from autoclaved estuarine sediments when mixed with a minimal medium, a phenomenon not observed without sediment (Ghoul *et al*., 1990). Gauthier and Rudulier (1990) found that addition of glycine betaine enhanced *E. coli* MC4100 survival (CFU) in three media (seawater, low- and high-organic carbon sediments) as well as uptake of radioactively labeled glycine betaine could be detected. The effect of glycine betaine on *E. coli* survival was clearly related to nutrient availability.

PREVIOUS ADAPTATION ON BACTERIAL SURVIVAL IN SEAWATER

Prior to arrival into the marine environment, enteric bacteria may have encountered wastewater, urban runoff, tidal flows, and other estuarine conditions of variable water quality and salinity common to coastal areas. They may be discharged directly from boats or bathers, remain for different periods in wastewater reservoirs, and/or are carried out to the sea through natural rivers via marshes or artificial conduits. Several studies demonstrate that enteric bacteria survival ability depends on their history preceding the seawater shock (Troussellier *et al*., 1998; Garcia-Lara *et al*., 1993; Gauthier *et al*., 1987, 1989a, 1989b, 1990, 1991 & 1992; Munro *et al*., 1987a, 1987b, 1989, & 1994), while some suggest that genetic adaptation may have enabled greater persistence in the marine environment. For example, while monitoring behavior of enteric bacteria in seawater with different previous wastewater exposure conditions, Dupray and Derrien (1995) assessed the ratio of culturable entero-bacteriaceae cells to the total population and found that loss of culturability of tested strains was smaller when bacteria had been in domestic wastewater before seawater. Previous exposure to sewage seemed to preserve or even induce some enzymatic activity, which was then maintained upon entering seawater--potentially an active metabolism enabling new protein synthesis. It seems that adaptation to drastic conditions occurred while in wastewater (lower temperature, lack of easily assimilated substrates, *etc*.), allowing for better survival in more drastic seawater conditions.

Similarly, Robeson and Skarmeta (1998) examined isolates of *E. coli* strains enriched from coastal waters impacted by sewage discharges in terms of growth response as a function of salinity and low temperature as well as patterns of resistance to anti-microbial agents (*i*.e. mercuric chloride, nalidixic acid, tetracycline, chloramphenicol, kanamycin, streptomycin, and ampicillin) and transmissibility of resistance markers. More than 50% of the strains showed salt tolerance and ability to grow at 7°C in contrast with control *E. coli* strains. About 15% displayed resistance to one or more antimicrobial agents and 6% transferred resistance markers in genetic

crosses including recipients of the genus Vibrio. Garcia-Lara *et al.* (1993) have shown that preadaptation by growth in an artificial seawater medium supplemented with complex organics (tryptone, peptone, and yeast extract) endowed the cells with greater viability in seawater. Munro *et al.* (1987a) demonstrated the same phenomenon using four enteric bacteria (*Shigella dysenteriae, E. coli, K. pneumoniae,* and *S. typhimurium*). But for four other strains, typically found in urban wastewater (*Staphylococcus aureus, S. faecalis, Pseudomonas aeruginosa,* and *Aeromonas hydrophila*), such pretreatment had no effect.

One dominant factor influencing seawater survival of bacteria is the growth phase during which the transfer to seawater occurred (Gauthier *et al*., 1992; Troussillier *et al*., 1998). Aerobically grown *E. coli* MC4100 cells exhibited high sensitivity to seawater (determined after two and six days of incubation) when exposed during the lag phase, lower sensitivity during the exponential phase and minimal during the stationary phase. When exposed to light, the difference between exponential and stationary phase cells was much less pronounced. For anaerobically grown cells, a different pattern was observed after one or two days of exposure, but not after six days.

In general, cells grown under anaerobic conditions were found to be much more sensitive than aerobically grown cells (Gauthier *et al*., 1989a & 1992), but this effect was not observed when TC was the parameter measured. Attrassi *et al.* (1996) found that anaerobically grown *E. coli* cell viability in seawater was 8.5 times lower than that of cells previously grown aerobically after two days of starvation in seawater and was 70 times lower for cells starved six days in seawater. Moreover, previous suspension of cells in sewage decreased three to nine times their subsequent viability after two days of starvation in seawater, but viability was no more affected after six days of starvation in seawater. This correlates well with the possible involvement of reactive oxygen species in the stress imposed by seawater, and hence with the significance of the adaptation to oxidative conditions.

In an effort to determine the nature of the protection acquired in the preadaptation process, *E. coli* MC4100 cells were exposed to five different types of stress for up to four hours; thermal (48 C), oxidative (1 mM H_2O_2), acid (pH 5.1), osmotic (0.5 M NaCl), and starvation (a nutrient-deficient buffer) followed by incubation in seawater (22 to 25°C) for six days. All pretreatments led to a better resistance to seawater compared to non-treated cells (Munro *et al*., 1994). Earlier, a 44°C pretreatment was found to decrease such resistance (CFU), as did growth at pH 6 or 8 when compared to pH 7 (Gauthier *et al*., 1989). Munro *et al*. (1994) concluded that the relative adaptive importance was osmotic, starvation, oxidative, acidic, and finally thermal. Apart from osmotic stress, all others induced higher resistance to seawater in an rpoS+ strain than in the

rpoS mutant, suggesting that the cross-protection endowed by oxidative, acidic, thermal and nutritional stresses was rpoS-dependent. Not surprisingly, these studies have also shown there is an increased resistance to seawater by cells preadapted to high osmolarity (*e.g.* 0.5 M NaCl or by 0.8 M saccharose). The preacquired capabilities for K^+ transport, glycine betaine synthesis or transport, and trehalose synthesis helped to increase *E. coli* ability to survive in seawater. Poorer resistance seems to result from hypoosmotic shock prior to seawater exposure (Gauthier *et al*., 1991; Combarro *et al*., 1992).

Another preadaptation variable tested was medium composition. Gauthier *et al.* (1989) reported significant differences between media only after more than seven days, but with no correlation to the richness or complexity of the medium. An additional adaptive parameter tested by Gauthier *et al.* (1990) was phosphate concentration; low phosphate grown cells, in which alkaline phosphatase synthesis was induced, were shown to remain culturable after longer periods in seawater. Most recently, Sinton *et al.* (2002) found that fecal coliform repair mechanisms appear to be activated in waste stabilization ponds such that the surviving cells exhibit greater sunlight resistance in natural waters than those from raw sewage. In contrast, Enterococcus appear to suffer photo-oxidative damage in the ponds, rendering them susceptible to further photo-oxidative damage after discharge, suggesting they are unsuitable as indicators of pond effluent discharges.

MOLECULAR MECHANISMS IN SURVIVAL OF ENTERIC BACTERIA

Rozen and Belkin (2001) underscore that many of the studies discussed to this point use colony formation and, to a lesser extent, other viability criteria to determine the genetic factors (*e.g.* rpoS) controlling sensitivity or survival of *E. coli* and other enteric bacteria in seawater. It appears that a substantial gap exists between these reports and the wealth of information presently available on bacterial physiology, biochemistry, and molecular biology. This gap is especially pronounced in the latter category; many of the molecular responses of *E. coli* and Salmonella to the environmental parameters mentioned above (salinity, starvation, pH, *etc*.) have been studied in great detail, but very few attempts were reported to make use of these data with respect to bacterial survival in seawater. They provide a detailed summary of these reports, including a table of all the genes potentially responsible for bacterial adaptation to seawater stressors.

SUMMARY AND CONCLUSIONS

The survival of enteric bacteria in the marine environment illustrates a set of attempts, many of them highly empirical in nature, to describe the actual fate of wastewater or enteric bacteria upon exposure to seawater. Only in recent years have these been expanded to include also some molecular aspects of the studied phenomena. The purposes of the review have been to summarize and clarify knowledge and derive some insight into the variability of enteric bacterial populations in coastal areas affected by urbanization.

While it is generally accepted that when enteric bacteria are exposed to seawater there is a loss of colony-formation ability on solid media, there is controversy related to the physiological state of these non-culturable cells. Bogosian *et al.* (1996 and 1998) claimed that VBNC cells are either dead or of no significance as they cannot be practically resuscitated. In contrast, other evidence suggests that they are not only viable but that pathogenicity may be retained (Pommepuy *et al*., 1996). Clearly, such an observation is of critical importance to the worldwide practice of releasing non-disinfected wastewaters into the sea and its potential public health consequences. The seawater-related VBNC controversy is a part of the broader issue (Kell *et al*., 1998; Barer and Harwood, 1999; Edwards, 2000) of the true meaning of various viability criteria and of the molecular and biochemical mechanisms controlling the shift from colony forming to VBNC. These regulatory systems are only partially understood, and probably vary according to the stimuli imposed on the cells.

When enteric bacteria are exposed to seawater, they are simultaneously challenged by a combination of stress factors, including high pH, low temperature, high salinity, limited nutrient availability, light radiation and its associated oxidative stress, and possible predation and competition. Light radiation and limited nutrient availability are probably the most significant stressors of this hostile combination, though potentially relevant only for shallow waters. Salinity appears to be less significant since when supplied with sufficient organic nutrients, *E. coli* can grow in seawater almost as well as it does in rich laboratory media while out-competing marine strains. Generally, lower pH and higher temperatures improve bacteria survival, as well as incorporation into marine sediments. Large hydrostatic pressures, relevant to deep-sea discharges of wastewater, have a variable effect on survival. Among possible biotic factors that may play a role in determining survival, predation by protozoa was the only one shown to be potentially significant. The contribution of other effectors, such as bacterial predation, viral infections, or unexplained antibiotic-like effects, was either insignificant or not demonstrated under natural conditions.

A recurring factor apparent from many studies is that in addition to actual seawater incubation conditions, previous growth history has a major influence on subsequent survival in the marine environment, suggesting significant adaptation potential. Such potential may play a role in recurring seawater contamination from stagnant waters or other coastal sources.

At the molecular level, the rpoS regulon is possibly the most significant among such adaptive systems. At least 50 different genes, under rpoS control, are induced by diverse stresses including salinity and starvation upon a shift to a stationary growth phase. Not all rpoS-controlled genes have been assigned a clear function, but among those that were, many combat effects of such stresses. The dominance of this regulatory mechanism was observed both in the significant negative effect that rpoS mutations had on *E. coli* survival and in the observation that rpoS-dominated genes accounted for 18 out of 22 shown to be induced by seawater exposure (Rozen *et al*., 2001).

Many of the studies considered here were motivated by the desire to protect near shore areas from fecal pollution and its ultimate effects on human health. In the design of marine sewage effluent outlets, bacterial die-off constants are often used (mostly *E. coli*). Such values may be based on both laboratory and field experiments, and it is of importance to recognize the limitation of both approaches. The significance of many of the field experiments is often site-specific, and they tend to ignore previous growth history of the monitored strains. Thus, mathematical models based on such results cannot replace the need for routine bacterial pollution monitoring. Conversely, laboratory experiments can be accurately designed to test the effects of specific parameters, but cannot simulate or imitate the complexity of the real marine environment. As such, additional information about seawater currents, temperature, and turbidity may be needed to complement indicator count measurements.

APPENDIX B DAILY FIELD FORMS

Included on supplemental compact disc

APPENDIX C DAILY FIELD SAMPLING FORMS

Included on supplemental compact disc

APPENDIX D SITE PHOTOGRAPHIC DOCUMENTATION

Included on supplemental compact disc

APPENDIX E CHAIN-OF-CUSTODY FORMS

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APPENDIX F THERMAL PLUME MODEL PARAMETERS AND OUTPUT

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APPENDIX G SIERRA ANALYTICAL REPORTS

Included on supplemental compact disc

APPENDIX H SCRIPPS ANALYTICAL REPORTS

Included on supplemental compact disc.

APPENDIX I SEVERN TRENT ANALYTICAL REPORTS

Included on supplemental compact disc

APPENDIX J STATISTICAL TABLES

Included on supplemental compact disc

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Included on supplemental compact disc